Australian Government

Department of Agriculture, Water and the Environment

Fire regimes that cause declines in biodiversity

Advice to the Minister for the Environment from the Threatened Species Scientific Committee on amendments to the List of Key Threatening Processes under the *Environment Protection and Biodiversity Conservation Act 1999*

Credit: © David Keith, Threatened Species Scientific Committee. Alpine ash (*Eucalyptus delegatensis*) in northern Kościuszko National Park 1 year on showing impacts of high-severity, high-frequency fires in 2019 and 2003.

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Fire regimes that cause declines in biodiversity include the full range of fire-related ecological processes that directly or indirectly cause persistent declines in the distribution, abundance, genetic diversity or function of species or ecological communities (see Figure 1). There is now strong scientific evidence that certain fire regimes threaten the persistence of much of Australia's biodiversity, even though it evolved through recurring fires over tens of millions of years. Particular fire regimes (combinations of fire frequency, intensity, season and type; Table 1), their landscape patterns and their interactions with other processes (e.g., drought, introduced species, disease, certain human activities) have been identified as a threat to more than 800 native species and 65 ecological communities listed as threatened under Australian legislation as at December 2020 (DAWE, 2021a). These numbers are likely to increase as further data emerge on the impacts of bushfires in 2019-20, making fire regimes that cause declines in biodiversity one of the most pervasive threats to Australia's biodiversity. The fire regimes that cause declines in biodiversity involve a diverse array of mechanisms. Indigenous people know many of them as 'wrong-way' fires, recognising the very complex intersection between fire regimes that cause declines in biodiversity with those that are inappropriate for culture and healthy country. While acknowledging the complexity of fire, culture and country, this Key Threatening Process listing describes which fire regimes cause declines of particular groups of species and ecological communities.

Fire regimes that cause declines in biodiversity vary across Australia between landscapes and climate types. The mechanisms that underpin fire-related threats are diverse - different fire regimes have been shown to disrupt life cycles or degrade habitats in various ways, depending on the characteristics of different species and ecological communities. How fire regimes threaten biota also varies across Australia between landscapes, habitats and climate types, and differ substantially between northern, central and southern Australia. Also, their impacts depend on contextual factors including drought, predation, herbivore activity, disease, and weed invasion.

While some fire regimes threaten species directly by reducing their survival and/or reproduction, many impacts of fire regimes on biodiversity are indirect, either because they alter habitats, disrupt dependencies among species, or exacerbate impacts of other threats. Fire regimes that threaten biodiversity may also degrade ecosystem functions, reducing the capacity of ecosystems to sustain native flora and fauna and to supply ecosystem services that support human well-being and livelihoods.

This Advice describes the suite of fire-related processes that threaten Australian biodiversity and demonstrates that 'Fire regimes that cause declines in biodiversity' is eligible for listing as a Key Threatening Process under the *Environment Protection and Biodiversity Conservation Act 1999* because it meets all 3 listing criteria:

- it could cause a native species or an ecological community to become eligible for listing in any category, other than conservation dependent;

- it could cause a listed threatened species or a listed threatened ecological community to become eligible to be listed in another category representing a higher degree of endangerment;

- it adversely affects 2 or more listed threatened species (other than conservation dependent species) or 2 or more listed threatened ecological communities.

Introduction

Fire regimes that cause declines in biodiversity include the full range of fire-related ecological processes that directly or indirectly cause persistent declines in the distribution, abundance, genetic diversity or function of species or ecological communities (see Figure 1). Declines in biodiversity and ecosystem function occur when fire regimes have direct effects on biotic and abiotic factors through heat or by-products such as smoke, particulates and noxious gases such as carbon monoxide. These direct effects may be attributable to, or are amplified by multiple components of fire regimes (frequency, intensity, season or type) or their spatial configuration or interactions with other processes. The consequent changes in ecosystems may alter or eliminate critical habitat components (such as soil structure and organic composition, shelter sites, food sources, foraging substrates, or breeding structures), keystone species, or energy and nutrient flows. Fire-related loss or marked decline of keystone species and structures or ecosystem engineers can result in extinction cascades, as dependencies among species are disrupted. Flammability feedbacks in some ecosystems may make it difficult to reverse the impacts of fire regimes that cause declines in biodiversity.

Fire regimes may also threaten biodiversity when they amplify the impacts of, or change the responses to, other environmental stressors. Examples include enhanced competitive abilities of weeds relative to native plant species, increased hunting efficiency of introduced predators, reduced evasion or defence by native prey species, increased impact of diseases or reduced disease resistance, as well as changes to the physical environment brought about by climate change, resource exploitation, species introductions, or land use change. For example, fire may promote the spread and increase the impact of significant pathogens such as *Phytophthora cinnamomi* (root rot disease) and increase the exposure and vulnerability of susceptible plants to myrtle rust.

A range of physical processes may reduce ecosystem resilience to fire, exacerbate fire-driven degradation of ecosystem structure or function, and elevate mortality or reduce reproduction in plant or animal populations. Examples of these processes include resource deprivation (for example, due to pre– or post-fire drought), changes to hydrology, habitat fragmentation, erosion and sedimentation (especially in aquatic systems), and physical disturbance (for example, associated with fire suppression operations). Although details of fire management *per se* are beyond the scope of the Key Threatening Process listing, it is acknowledged that fire management activities by humans are diverse, and may have positive or negative effects on biodiversity, depending on their context and methods of application.

Fire regimes may threaten the persistence of species across a wide taxonomic spectrum (e.g., plants, vertebrates, invertebrates, fungi) and a diverse range of ecosystems, including those that are naturally fire prone (e.g., heathlands, sclerophyll forests) and those that rarely burn (e.g., rainforests, alpine ecosystems).

1   Name and description of the threatening process

1.1   Title of the process

Fire regimes that cause declines in biodiversity.

1.2   Description of the process

Fire regimes that cause declines in biodiversity include the full range of fire-related ecological processes that directly or indirectly cause persistent declines in the distribution, abundance, genetic diversity or function of a species or ecological community (see Figure 1 of this Advice). Under the *Environment Protection and Biodiversity Conservation* (EPBC) *Act 1999* (s528), biodiversity means the variability among living organisms from all sources (including terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part) and includes:

(a) diversity within species and between species; and

(b) diversity of ecosystems.

In simple terms, biodiversity encompasses the full variety of life, encompassing genes, species and ecosystems. Fire is a natural ecological process that interacts with the evolutionary and climatic history of landscapes (Bowman et al., 2009). Thus, the continental and local distribution and abundance of the biota has been influenced by the intensity, frequency, season, and type of fire, and the size and spatial configuration of successive fire events (Crisp et al., 2011; Lynch et al., 2007). Fire regimes have had a profound influence on Australia's biodiversity over tens of millions of years (Lamont et al., 2019), including the period since the arrival and establishment of humans on the continent more than 50,000 years ago (Bowman et al., 2009). Extensive transformation and fragmentation of Australian ecosystems by contemporary human activity, altered ignition patterns and anthropogenic climate change are causing major alterations to the frequency, severity and seasonal pattern of fires, and their extent and spatial configuration across landscapes.

Despite the long evolutionary history of fire in Australia, altered fire regimes, in combination with increased sensitivities to the new regimes and global climate change, are driving declines and loss of biodiversity and ecosystem function. Certain types of fire regimes threaten biodiversity by disrupting species' life cycle processes, altering habitats, disrupting dependencies among species or exacerbating impacts of other threats. As at December 2020, fire regimes are identified as a threat to more than 800 species and 65 ecological communities listed as Critically Endangered, Endangered or Vulnerable under the EPBC Act (DAWE, 2021a; 2021b). This Advice addresses the fire-related threats to biodiversity across Australia's terrestrial environments and downstream impacts on freshwater environments and, potentially coasts and marine environments (Ley & Chester, 2020; Li et al., 2021; Manjalay, 2020; Silva et al., 2020). These fire-related threats are typically a consequence of human activity, including deliberate or neglectful management of fire.

Fires are significant disturbance events in almost all Australian terrestrial ecosystems (Keith & Tozer, 2017), and play a vital role in shaping and maintaining biodiversity patterns and processes (Bradstock et al., 2002; 2012). As a mechanism of ecosystem disturbance, fire can maintain or enhance biodiversity, or cause the loss of biodiversity (Fox, 1982; Keith, 1996; Whelan et al., 2006; Williams et al., 2002; Yates et al., 2008). Manipulation of fire regimes has a long history in Indigenous Australian culture (Fletcher et al., 2021), and remains the main landscape management tool in many terrestrial environments and most conservation reserves (Woinarski, 1999). Many Australian species and ecosystems demonstrate a level of fire resilience, with some being dependent on, or limited by, particular fire regimes (Bradstock et al., 2002; Hammill et al., 2016; Miller et al., 2017). Consequently, fire management activities may have positive or negative effects on biodiversity, depending on their context and methods of application.

Changes in fire regimes over tens of millions of years and strong regional variation across the continent reflect a dominant influence of climate through its effect on: i) seasonality, frequency and severity of fire weather conditions; ii) the moisture content of biomass and hence its ability to burn; iii) the structure of vegetation and hence the volume and structure of fuels; and iv) ignition patterns, such as the incidence of dry lightning (Abram et al., 2021; Bradstock, 2010; Dowdy, 2020). The dependency of these causal factors on the prevailing climate highlights the fundamental sensitivity of fire regimes to global climate change (Bowman et al., 2020).

Since their arrival in Australia in recent geological time, humans have influenced fire regimes, particularly through ignition patterns and alterations to vegetation structure that, in turn, change fire regimes. Since European settlement, human influence on fire regimes again changed markedly across almost all parts of Australia (Fletcher et al., 2020; Mariani et al., 2022; Mooney et al., 2011; Russell-Smith et al., 2007). Landscape modifications by humans, such as through vegetation clearing, timber harvesting or livestock grazing, extinction of native species and the proliferation of alien plant species, have caused changes in fire behaviour and have altered the resilience of native biota and their habitats to fire (Bowman et al., 2016; Russell-Smith, 2001). In this document the term 'alien species' refers to species that have been introduced to Australia from other countries, as well as species native to Australia that now occur outside their pre-European range.

Particular fire regimes have been identified as an important threat to some components of biodiversity (for example, Bradshaw et al., 2018; Shedley et al., 2018; Threatened Species Scientific Committee, 2011), but until now there has been no systematic evaluation of interactions between fire regimes and biodiversity across all Australian taxa and ecosystems. This Advice aims to provide a synthesis of fire-related processes that threaten biodiversity in different ways across Australia. Some of these processes threaten species directly by reducing their rates of survival and/or reproduction, but most impacts of fire regimes on biodiversity operate indirectly, either by reducing the suitability of habitats, disrupting dependencies among species, or exacerbating impacts of other threats. Fire regimes that threaten biodiversity may also degrade ecosystem functions, further reducing their capacity to sustain native flora and fauna, and eroding their capacity to deliver ecosystem services that support human well-being and livelihoods (Russell-Smith et al., 2009). This synthesis draws from a substantial body of scientific and traditional knowledge on fire and biodiversity conservation in Australian landscapes (Andersen et al., 2003; Bradshaw et al., 2018; Bradstock et al., 2002; 2012; Burrows & Abbott, 2003; Cary et al., 2003; Driscoll et al., 2016; Gill, 1975; Gill et al., 1981; 1999;

Keith, 1996; Pike, 2008; Russell-Smith et al., 2009; 2019; Whelan, 1995; Woinarski, 1999; Woinarski & Legge, 2013).

1.3   Name changes

A public nomination was received to list 'Contemporary fire regimes resulting in the loss of vegetation heterogeneity and biodiversity in Northern Australia' as a key threatening process. As many issues raised in the original nomination are also relevant in other areas of Australia, the name and concept of the Key Threatening Process was amended to 'Fire regimes that cause declines in biodiversity' to accommodate all species and ecological communities affected.

2 The changing fire landscape

2.1 What is a fire regime?

The core concept of a 'fire regime' refers to the frequency, intensity, season, and types (aerial/subterranean) of successive fire events at a point in the landscape (Table 1; Gill, 1975). Each point in a landscape therefore has a unique fire regime (defined by the four core components; Gill, 1975), enabling landscape-level patterns in fire regimes to be described spatially as fire mosaics (Bradstock et al., 2005) with relevant spatial metrics (Table 1).

In addition to the four core components of fire regimes, there are several related fire characteristics that must be considered in order to understand the ecological effects of fires (Table 1). One of these is 'fire severity' which refers to the amount of organic matter consumed by a fire (Keeley, 2009), as opposed to 'fire intensity' which refers to the amount of energy released during a fire (Alexander & Cruz, 2012; Burrows, 1999; Cheney, 1990). Although intensity and severity are broadly related, they are not always aligned, and sometimes major ecological effects associated with high fire severity can result from low-intensity fires (e.g., due to prolonged smouldering) and *vice versa* (Keeley, 2009). As fire severity is *generally* more relevant to ecological effects (and hence biodiversity conservation) than fire intensity, the focus of this Advice is primarily on the former concept.

Table 1 Components of the fire regime and related descriptors

Fire regime descriptor	Definition	Example metrics	Related variables
Core components: Fire Frequency	The number of fires per unit time at a point in the landscape.	Fires per decade; probability of burning at a point. Inversely related to mean fire interval. Recent fire frequency is indicated by length of the most recent fire interval.	Time since last fire; Mean and minimum fire interval over a specified time frame; Fire history (number of fires per unit time for a specified area; not necessarily overlapping).
Core components: Fire Intensity	Energy output or heat release at a point in the landscape during a specified fire event.	Time-averaged energy flux (Watts per m^2). Also Fireline Intensity (per metre of fire front) = H.w.R, (H = energy content of fuel, w = biomass consumed, R = rate of spread).	Severity (and associated measures); Flame length (correlates of intensity); Mean and variation in intensity over multiple events in a specified area or time frame.
Core components: Fire Season	The time of year for a specified fire event.	Julian days (quantitative), month, season	Mean and variation in season over multiple events in a specified area or time.
Core components: Fire Type	Whether a specified fire event at a point in the landscape burns at or above ground level (live/dead biomass), or below the surface (semi-decomposed organic matter, for example, peat, coal).	Surface or substrate fire (binary descriptor), noting that some fires may consume both substrate and above-ground biomass.	Fire severity (see below in this table for metrics of above- and below-ground fire severity).

Fire regime descriptor	Definition	Example metrics	Related variables
Related features: Fire severity	The amount of organic matter consumed in a fire event at a point in the landscape.	Percent crown volume scorch (for each vertical vegetation stratum); Minimum terminal twig diameter; Spectral indices (for example, Normalised Burn Ratio); Depth and surface cover of peat loss (substrate fires – see Fire Type).	Fire intensity
Related features: Fire extent	Area within the spatial boundary of a fire event.	Estimates may be based on remote sensing, ground traverse or both. The degree to which mapped fire extent includes unburnt patches depends on map resolution and methods.	Landscape fire scars, percent landscape burnt for a specified area.
Related features: Fire patchiness	The spatial configuration of patches with different fire characteristics within a specified area.	Area statistics and patch size, shape, edge length and separation metrics for areas burnt at different levels of severity and areas unburnt within a fire footprint.	Fire mosaic, Fire extent, Fire severity.
Related features: Ignition type	The causes of fire ignition.	Statistics on fire causes: lightning; managed ground ignitions; spot ignitions, line ignitions, managed aerial ignitions, accidental ignitions, arson, and so forth.	Fire patchiness.

Note: The four core components (frequency, intensity, season, type) describe fire characteristics at a point in the landscape. The point concept (Gill, 1975) enables them to be aggregated spatially to describe fire mosaics using other metrics. Indigenous Australian calendars of seasons, which vary around Australia (e.g., Garde, 2009), otherwise described as spring/summer/autumn/winter, or in the wet-dry tropics dry season and wet season. Terms describing patchwork fire patterns exist in some Indigenous Australian languages (e.g., Garde, 2009).

2.2   Fire regimes, past, present and future

Fire regimes are dynamic. They have changed over extended geological and recent human time scales and will continue to change into the future. Geographic and long-term variations in climate have proven to be a critical driver of change in fire regimes (Abram et al., 2021; Dowdy, 2018; Harris & Lucas 2019; Van Oldenborgh et al., 2021). To provide context to fire regimes as a contemporary Key Threatening Process, sections 2.2.1 Pre-Human, 2.2.2 First Nations, 2.2.3 Post-European colonisation, and 2.2.4 Anthropogenic drivers of future trends summarise the current understanding of fire regimes, respectively, during four eras.

2.2.1   Pre-Human

Fire has a long history on the Australian continent, extending beyond tens of millions of years (Bowman, 2000; Martin, 2006). The evidence of this extended history of fire includes fossil charcoal or fusain and inferred evolutionary responses of Australia's biota shown in correlative phylogenetic studies of fire-related traits such as serotiny that emerged in plants in the early Tertiary period, between 60 and 100 million years ago (He et al., 2011). Fire had an increasing influence on Australian vegetation from the Oligocene – Miocene boundary (~23 Ma) onwards,

promoting the expansion of sclerophyllous taxa and graminoid plants. In tandem with a drying climate, increased fire is implicated in the contraction of rainforest taxa and ecosystems to the eastern ranges (Bowman, 2000; Kershaw et al., 2002). Enhanced fire activity was associated with continental drying trends that fluctuated through to the present. These climate-driven changes to fire regimes have profoundly influenced the evolution and distribution of Australia's biota (Bradshaw et al., 2011; Causley et al., 2016; Crisp et al., 2011; Hill et al., 2016; Midgley & Bond, 2011). This extended period of fire activity pre-dated the evolution of humans by tens of millions of years.

2.2.2   First Nations

Since their arrival on the Australian land mass more than 50,000 years ago, Indigenous Australians are likely to have influenced fire regimes directly through ignition behaviour, and other factors, and indirectly through any effects of their fires on vegetation structure and composition. There is evidence that use of fire varied between regions and is unlikely to have remained constant in any given area (Hiscock, 2008). For example, valley fill in the Sydney region during the late Holocene (3,000 - 4,000 years ago) cannot readily be attributed to climate processes, with changes in burning activity identified as one plausible explanation (Kohen, 1996).

The limited archaeological record, as well as the disruption of culture and loss of many oral histories, pose substantial challenges to reconstructing regional histories of fire regimes during the First Nations era and prior to European colonisation, especially in southern Australia, despite persistence of a strong contemporary Indigenous culture. Cultures, climate, landscapes and fuels also vary markedly across the country, and so inferences drawn from one region may not reflect practices in other regions. Given the dynamic nature of human cultures, practices documented by early settlers and ongoing practices of contemporary Indigenous land managers likely exhibit both similarities and differences compared to those of earlier historical times (Perry et al., 2018).

Despite uncertainties about how patterns of fire use varied between regions and changed through time (Johnson, 2016), there is abundant evidence that Indigenous Australians used landscape fires for a variety of purposes, in many regions and over extended time. Their motivations for influencing fire regimes are framed by spiritual beliefs and cultural customs and inextricably linked to resource and access management, notably promoting fire-responsive food plants and habitats for prey and totem animals (Bliege Bird et al., 2020; Garde, 2009; McGregor et al., 2010; Pike, 2008). Successful outcomes depend on decisions about where and when to ignite fires, and thus influence the frequency, severity, season and spatial patterns of the resulting fire regimes (Table 1). Conflicts, experimentation, errors and other fundamental aspects of human behaviour may also have influenced fire regimes of the First Nations era, as they continue to do today.

Fire regimes during this era were, and still are, influenced by, and responsive to weather and climate (e.g., Bliege Bird et al., 2016). Weather conditions impose constraints on when fuels in humid-subhumid climates are sufficiently dry, and when grass fuels in dry climates are sufficiently connected, to enable fire propagation, in turn influencing seasonal and interannual timing and patterns of human-ignited fires. At various times through the fossil charcoal record, during and prior to human occupation of Australia, there is evidence of major fire events (Atahan et al., 2004; Mooney et al., 2011; Wright et al., 2021a).

2.2.3   Post-European colonisation

In many regions, including large areas of tropical and arid Australia, Indigenous fire management remains, and is a means of restoring a fundamental expression of Indigenous knowledge and connection to local ecology and country (McGregor et al., 2010; Russell-Smith et al., 2009). The widespread transition from Indigenous to European fire regimes that occurred in the nineteenth and twentieth centuries varied between ecosystems as different land uses developed (Bliege Bird et al., 2013; Bowman, 1998; Bowman & Panton, 1993). A driver of change in anthropogenic fire regimes has been the depopulation of Aboriginal people from their traditional lands or the interruption of traditional fire practices, a transition that occurred at different times around Australia (Blackwood et al., 2021; Burrows & Chapman, 2018; Burrows & Christensen, 1990; Burrows et al., 2006; Russell-Smith et al., 2003). This may have resulted in significant changes to vegetation. For example, in south-eastern Australia, Mariani et al. (2022) associated disruption of Aboriginal land management with increased shrub cover and less grass in forest and woodland environments.

With some exceptions, the broad trend in charcoal and fire scar evidence, largely from south-eastern Australia, suggests increased fire activity over much of the continent, from the mid-19th until the mid-20th century (Mooney et al., 2011). Since contemporaneous fire mapping began to improve in the 1970s, with the advent of satellite imagery, there also appears to be an increase in the frequency and extent of both unplanned and planned fires, at least in some regions (e.g., Lindenmayer & Taylor, 2020).

The spatial patterns of fire have changed markedly since European colonisation; for example, the eastern, southeastern and southwestern seaboards have experienced habitat fragmentation following agricultural development, the embedding of large human populations and associated infrastructure within flammable forests, and increasingly widespread prescribed burning conducted across production forests and conservation areas (DBCA, 2020; Inspector-General for Emergency Management, 2015).

Fire in the tropical savannas includes a dominant component of human ignitions (Russell-Smith et al., 2007). There is now an increasing move to reduce the occurrence of extensive high-severity mid to late dry season savanna fires by igniting lower-severity early dry season fires (Russell-Smith & Edwards, 2006). These early dry season fires tend to be patchier at both macro and micro scales (Edwards et al., 2021; Legge et al., 2011; Russell-Smith et al., 2015; Werner, 2010; Wysong et al., 2022). In some areas, burning by graziers still occurs across portions of the savanna landscape each year, to protect high-quality pastures from fires originating in lower quality pasture areas, to produce nutritious 'green pick' for cattle during the dry season, and to facilitate mustering; in other areas graziers attempt to exclude fire to maximise pasture for their stock (Lewis, 1985; Skroblin et al., 2014). In the eastern Australian subtropics, graziers burn to prevent the return of trees to their grazing lands (Fensham, 2012).

Rapid climate change has become a major threat to biodiversity in recent decades and has already caused major shifts in fire weather in southern Australia that have significant implications for fire regimes, as detailed in section 3.3. For example, more temperate forest was burnt in the 2019-20 fire season than previously recorded (Boer et al., 2020). The recent intensity (energy output) and frequency of extreme pyroconvective events (e.g., fire-generated thunderstorms) is unmatched in the historic record (Abram et al., 2021). Areas such as the

Gondwanan rainforests that have seen little fire in centuries to millennia have become more exposed to fire (Bowman et al., 2019; Harris et al., 2018; Kooyman et al., 2020).

2.2.4   Anthropogenic drivers of future trends

Climate change is likely to be a primary anthropogenic driver of future trends in fire regimes (section 3.3), however, other human actions are altering both the rate of ignitions and the flammability of the landscape. Interactions with anthropogenic drivers such as climate change, land use, and biotic invasions are already transforming fire activity and its impacts on biodiversity, with effects forecast to extend over the coming decades (Kelly et al., 2020).

The occurrence of fires in seasons with less lightning has been interpreted to indicate predominantly anthropogenic ignition sources including in the more tropical regions of Australia (Russell-Smith et al., 2007). In the more temperate regions of southern Australia lightning-ignited fires account for a considerable proportion of the area burnt by all fires (e.g., around 50% or more in Victoria; Dowdy & Mills, 2012). Climatic changes are increasing the number of lightning ignitions in some temperate southern areas of Australia (Bates et al., 2018; Mariani et al., 2018), and changes in human populations and industries that cause ignitions will have long-term effects. The frequency of ignitions is expected to increase in coastal areas, for example, due to increasing population density (Collins et al., 2015). Climate change is also affecting the abundance and properties of bushfire fuels, including their dryness and availability to burn. In the Great Victoria Desert, annual rainfall has increased by 20-30 mm per decade in recent times (Pianka & Goodyear, 2012), with associated changes in shrub-grass composition and fire regimes (Burrows & Chapman, 2018). The mechanisms underpinning these changes require further research.

Some ecosystems have been made more fire-prone due to increased disturbance rates. Several studies have reported that forestry operations initially increase the likelihood of high-severity fire in temperate areas because a more open forest canopy increases exposure of understorey vegetation to high rates of fuel drying during dry periods, amongst other effects (Furlaud et al., 2021; Lindenmayer et al., 2021; Lindenmayer & Taylor, 2020; Taylor et al., 2014). However, this effect is conditional on fuel moisture and fire weather, which have stronger effects than forestry on the extent and spatial patterns of fire severity (Bowman et al., 2021a; Taylor et al., 2021a). It should be noted that forestry activities occur in a relatively small proportion of the landscape and are not related to ignitions, which initiate many of the largest fires (Bowman et al., 2021a; Keenan et al., 2021). Increases in fire frequency promote further fire in some forest and woodland ecosystems ('landscape traps', Lindenmayer et al., 2011), with elevated risks lasting for decades to centuries (Dixon et al., 2018; Gosper et al., 2013; Zylstra, 2018b).

In an effort to mitigate fire threats, prescribed fire has been used for decades in south-western Australia, reportedly leading to reductions in the extent of wildfires (Boer et al., 2009). While some prescribed fires are locally targeted, broadscale prescribed fire has been used across remote areas (Inspector-General for Emergency Management, 2015), at times introducing fire to large areas of very long-unburnt ecosystems (Bradshaw et al., 2018).

Livestock grazing appears to have varying effects on landscape flammability, with overgrazing limiting grass biomass in some areas and promoting replacement of large shrubs with grasses (thus increasing flammability) in others (Connor et al., 2018).

Legacies of fragmentation and weed invasion are also driving changes in fire regimes. Fragments of woodland or forest in rural or urban landscapes are burning less frequently due to reduced fire connectivity, and some are undergoing invasions by less flammable plants (e.g., *Pittosporum undulatum*; Gleadow & Walker, 2014). Fire frequency and/or severity have increased in areas where highly flammable grasses, such as the invasive environmental weed gamba grass (*Andropogon gayanus*), have spread through savanna (Russell-Smith et al., 2019; Setterfield et al., 2010). Other invasive species, particularly buffel grass (*Cenchrus ciliaris*), are also increasing the spatial extent, frequency and severity of fires in arid areas (Schlesinger et al., 2013; Wright et al., 2021b).

3 Mechanisms of biodiversity loss

Fire regimes that cause declines in biodiversity involve a diverse array of ecological mechanisms that interact with one another in complex ways. The processes elevate the risks of population decline or extinction and ecosystem collapse in large numbers of listed and non-listed species and ecological communities across Australia. A clear exposition of these mechanisms, their dependencies, and expression is essential to inform effective management and recovery strategies to reduce the risks. Fire-related threats operate by either directly affecting aspects of an organism's life cycle, or by indirectly affecting ecological interactions and other processes (Figure 1). The outcomes include declines in species' populations, erosion of genetic diversity and adaptive capacity and loss of ecosystem function.

Figure 1 Individual species and ecological communities may be exposed to one or more fire-related mechanisms of threat
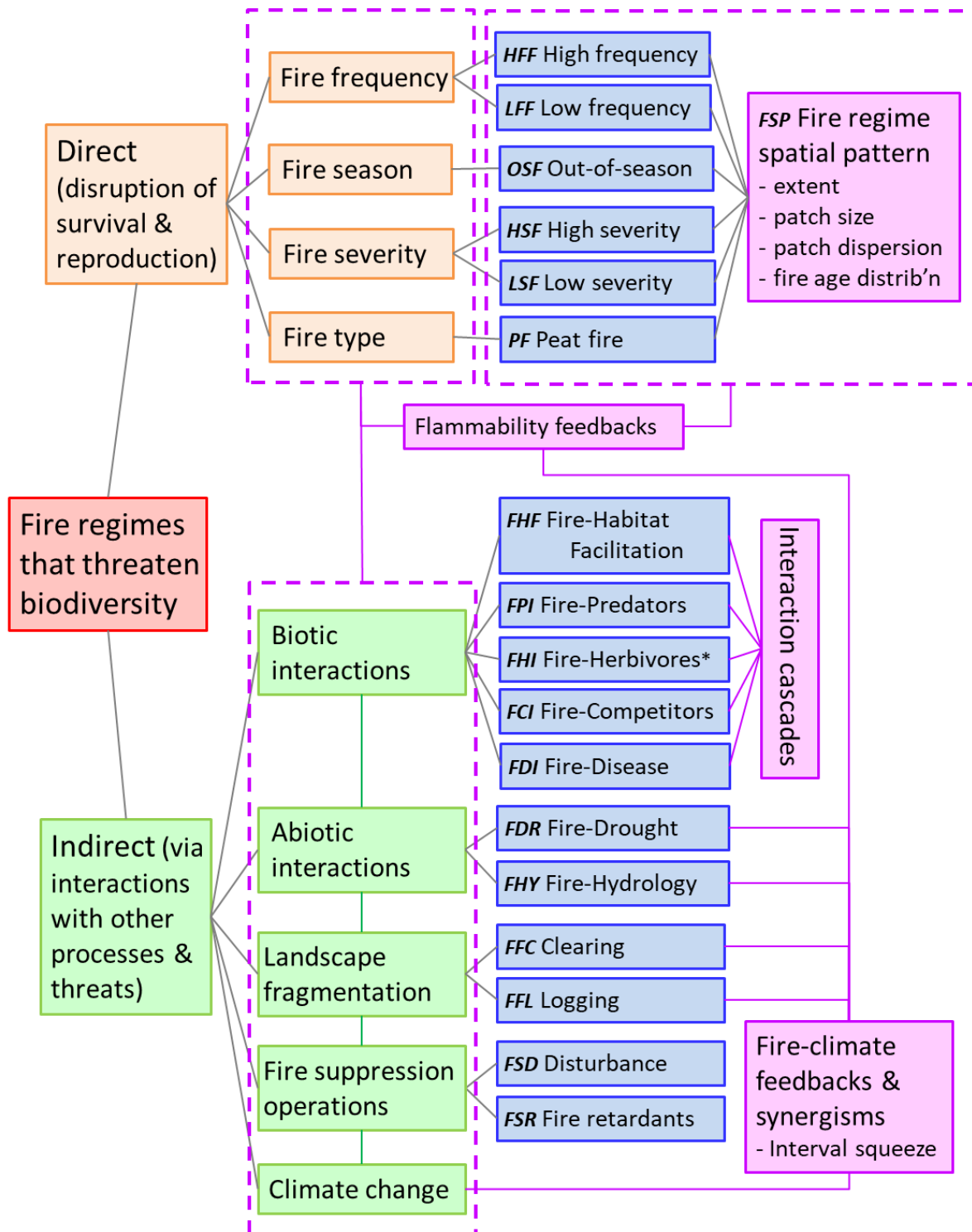
Source: Threatened Species Scientific Committee

Fire severity is used to describe fire-related threats throughout this Advice because the concept of severity is more relevant to ecological effects of fire than 'intensity' (a concept used to describe the physics of fire behaviour). The mechanisms in Figure 1 may be directly related to disruption of life cycle processes (survival, reproduction) by components of the fire regime (HFF High frequency, LFF Low frequency, OSF Out-of-season, HSF High severity, LSF Low severity, PF Peat fire) or their spatial patterns (FSP) or they may operate indirectly through fire-mediated disruptions to ecological interactions and amplification of other threatening processes. Indirect mechanisms include interactions between fire and several biotic processes, including effects on habitat suitability (FHF Fire-habitat facilitation, FPI Fire-predators, FHI Fire-herbivores, FCI

Fire-competitors, FDI Fire- disease), as well as interactions between fire and abiotic processes that limit life cycle processes or degrade habitats (FDR Fire-drought, FHY Fire-hydrology). Other threats that interact with, or depend on fire include those associated with landscape fragmentation (FFC Clearing, FFL Logging) and those associated with fire suppression operations or infrastructure development (FSD Disturbance, FSR Fire retardants). Secondary interactions among fire and multiple biotic processes may exacerbate threats through complex interaction networks and cascades, which are poorly studied. Flammability feedbacks may occur when certain fire regimes increase flammability of certain ecosystems. Climate change may amplify or alter responses to direct and indirect fire-related mechanisms of threat, and may also generate feedbacks.

The wide range of species and ecological communities threatened by different combinations of fire-related mechanisms is illustrated by a suite of examples in Table 2. The evidence implicates multiple realised and potential mechanisms underlying risks to most of the species and communities. Many of these are already listed as threatened under the EPBC Act 1999, while others are likely to become eligible for listing if fire-related threats continue operating.

3.1 Threats posed directly by components of the fire regime

The four core components of the fire regime (frequency, season, intensity and type) may threaten biodiversity, individually or in combination and through their extent and spatial pattern in landscapes. While 'fire intensity' is a concept used to describe the physics of fire behaviour, this Advice uses the related concept 'fire severity' to describe fire-related threats to biodiversity because the concept of severity is more relevant to ecological effects of fire (see Table 1 for definitions of both terms). Certain combinations of these components disrupt survival and reproduction, the vital life cycle processes that sustain species populations (Keith, 2012; Whelan, 1995). Ecosystems and ecological communities may also be threatened where these direct fire-related threats drive population declines in species that have key functional roles. Interactions among different components of the fire regime and their spatial pattern appear to be common, making it difficult to isolate individual causal factors. For example, in temperate and tropical Australia, fire severity is linked to fire season, with severe fires more likely to occur during summer in the south and the mid-late dry season in the north (Murphy et al., 2013). Their effects may also be linked. For example, whether severe fires drive declines in populations of arboreal marsupials in mountain ash forests depends on their extent and frequency of recurrence in relation to rates of tree maturation and development, seedbank accumulation and animal movement (Ashton, 1981; Bowman et al., 2014; Lindenmayer, et al., 2021). Nonetheless, it is possible to identify how particular characteristics of fire regimes threaten particular components of biodiversity, despite the limited biological information available on fire responses of many threatened species and ecological communities currently listed under the EPBC Act.

3.1.1 Fire frequency (High frequency, HFF; Low frequency, LFF)

Fire affects many species immediately through event-related mortality or tissue damage and indirectly though sudden changes to habitat structure (see section 3.2.1.1 Fire effects on habitat suitability and facilitation interactions (FHF)). If populations are to persist through a sequence of fires, there must be sufficient time between successive events to enable population recovery through reproduction or immigration of new individuals, as well as recovery of habitat suitability for species that depend on resources that become scarce after fire. Subsequent fires may disrupt these recovery processes causing population declines or extinctions if fire intervals are short (i.e. fire frequency is high) relative to the timing of life cycle processes (Keith, 2012). In some species, these recovery processes occur rapidly, while others have inherently slow rates of growth, maturation, reproduction or movement, or are contingent on slow recovery of suitable habitats. In many areas, increasing fire frequency has direct impacts on the diversity and

abundance of vertebrate, invertebrate, plant and fungal populations (Figure 1, mechanism HFF). These effects are likely to be exacerbated by drying climates (FDR) particularly in temperate Australia (Enright et al., 2015).

For plants, high fire frequency may pose a threat to both obligate-seedling and resprouting life-history types. For 'obligate seeders', standing plants are killed by fire above a certain severity, and population persistence depends entirely on recruitment from seeds. Seed banks stored in the soil (geospory) or in plant canopies (serotiny) are the source of recruitment after fire in many of these species and require a minimum time between successive fires for replenishment. Hence, short fire intervals disrupt their life cycles and cause population declines (for example, in the **Vulnerable** *Banksia verticillata* (Granite Banksia), Threatened Species Scientific Committee, 2008). The minimum fire intervals that ensure population persistence vary among species within an ecosystem, between different ecosystems and with environmental conditions. Some species accumulate substantial seed banks within a few years of fire, whereas the slowest seed bank accumulators require two or more decades (Fairman et al., 2016), even if seed release or germination is stimulated by fire. Resprouting woody plant species can also be vulnerable to a fire frequency that is too high to permit replenishment of resources or structures to support post-fire regenerative responses (Enright et al., 2011), although the effects may be compensated by seedling recruitment that replaces mature plants over time. A number of resprouting woody plants that are slow to develop fire-resistant or regenerative organs, such as lignotubers or thick bark, are also vulnerable to high fire frequency (HFF) because attritional mortality of standing plants cannot be counter-balanced by recruitment, especially when fecundity is low (Keith, 2012; Tozer & Keith, 2012). Other plant species lack both the capacity to resprout and persistent seed banks (e.g., mistletoes), and these may also be threatened by high fire frequency, depending on rates of dispersal and establishment (Gosper & Prober, 2020). Some vertebrate animals are also threatened by high fire frequency, although the mechanisms typically involve indirect processes as well (see section 3.2.1.1 Fire effects on habitat suitability and facilitation interactions (FHF)), either as a cause of mortality or as a disruptor of compensatory recruitment or immigration (Santos et al., In review). Shrub-nesting birds, for example, depend on post-fire re-establishment of substrates that do not begin to become suitable for nesting until at least 6 years after fire (Brooker & Rowley, 1991). Avoidance of population declines in those species may require fire intervals longer than 10 years to enable multiple breeding seasons between successive fires (Keith et al., 2002) and likely longer under climate change or in degraded ecosystems.

The ecological communities most threatened by HFF include those with obligate-seeding plants as structural or functional dominants, such as the **Endangered** Arnhem Plateau Sandstone Shrubland Complex, Eastern Stirling Range Montane Heath and Thicket and currently unlisted Alpine ash forests in the Australian Alps bioregion (Table 2), and those prone to slow recovery from structural transformation, such as **Critically Endangered** Lowland Rainforest of Subtropical Australia (Table 2). Desert ecosystems dominated by obligate-seeding plants, notably an array of mulga communities, may also be threatened by HFF, including where increases in the abundance of flammable grassy fuels are driven by invasive species (see section 3.2.1.4 Fire effects on competitive interactions (FCI)) and changes in rainfall patterns (see section 3.3 Climate change) (Leavesley et al., 2010; Nano et al., 2017).

Low fire frequency (LFF, Figure 1) may also directly cause declines in species' populations and ecological communities by failing to trigger essential life-history cues, through autogenic

changes to habitat suitability, or through interactions with other threats such as fragmentation (see section 3.2.2.3 Fire-fragmentation effects through vegetation clearing (FFC), Figure 1). Evidence of threats posed by long fire intervals comes from relatively few cases (Table 2). Traits that typically make plant species sensitive to long intervals between fires or other suitable disturbances include short-lived seed banks, fire-cued seed germination, an inability of seedlings to establish in unburnt conditions, low seed-dispersal range, and declining fecundity a few years after recruitment. Some of these traits appear to be related to putative declines in the **Endangered** *Verticordia fimbrilepis* subsp. *fimbrilepis* (Shy Featherflower) (Table 2). The natural range of this species has been highly fragmented by land-clearing, which has limited fire spread, resulting in low fire frequencies (Yates & Ladd, 2010). Animals are affected indirectly through mechanisms of vegetation change and associated resources which become scarce with increasing time since fire (see section 3.2.1.1 Fire effects on habitat suitability and facilitation interactions (FHF), Figure 1).

Certain critical ecosystem components have been maintained by frequent fire in alternative stable states ('plagioclimax communities'), apparently for millennia. They could revert to another stable state through autogenic succession, given sufficiently long fire intervals. 'Grassy balds' found within forested landscapes in subtropical eastern Australia to Tasmania were likely maintained in this way (Fletcher et al., 2020), and some may now be contracting as a result of encroachment by surrounding woody vegetation. The historical extent of such features is often obscured by more intense and widespread post-colonisation land use and other disturbances, and evidence for their maintenance by human-ignited fires is indirect, but nonetheless plausible (Butler et al., 2014). Declining availability of grass seed and increased susceptibility to predators due to encroachment of *Melaleuca* shrubs into savanna is one of several threats to the **Endangered** *Psephotus chrysopterygius* (Alwal, Golden-shouldered Parrot) (Table 2) (Garnett & Crowley, 2002). The encroachment process occurs under low fire frequencies, and is likely promoted by legacies of cattle grazing, which reduces grass seeding and biomass and alters soil structure in favour of shrubs (see section 3.2.1.3 Fire effects on herbivore-plant interactions (FHI), Figure 1). Some temperate heathlands or swamps such as the **Endangered** Coastal Upland Swamps in the Sydney Basin Bioregion may transition to thickets under low frequency fire regimes (Table 2), while hydrological factors (see section 3.2.2.2 Fire-hydrological interactions (FHY)) that may condition the transition are yet to be fully resolved (Keith et al., 2007; Mason et al., 2017).

3.1.2   Fire season (Out-of-season, OSF)

The season in which fire occurs can affect plants and animals directly by disrupting phenological processes, or indirectly by affecting other species that influence habitat, food, trophic interactions or mutualisms such as pollination (Figure 1, mechanism OSF, out-of-season fires).

Fire season affects the population viability of plants through mechanisms influencing the survival of adult plants, the availability of propagules, and the post-fire establishment of seedlings (Miller et al., 2019; Roche et al., 1998), as well as dispersal of plant propagules (Keith et al., 2020a). Resprouting plants, for example, require sufficient non-structural carbohydrates to recover from disturbance (Clarke et al., 2013), so if fire occurs at a time of year when such resources are at low levels, resprouting may be inhibited, reducing survival rates. Terrestrial orchids in temperate Australia may be especially sensitive to this mechanism, as out-of-season fires in autumn to spring (during the growth season) remove aerial shoots after starch reserves in tubers are deployed in above-ground biomass and before they may be fully replenished,

whereas these risks are avoided when fires burn during the dormant phase in summer, the 'natural' season of fire in temperate latitudes (Jasinge et al., 2018a; 2018b).

Plants with physiological seed dormancy mechanisms, such as the **Vulnerable** *Leucopogon exolasius* (Woronora Beard-heath), are also sensitive to fire season (Table 2). Cool season fires generate delayed and reduced levels of seedling emergence that are less likely to counter-balance the loss of fire-killed adults compared to the higher rates of recruitment that occur after summer fires (Ooi et al., 2007). Similarly reduced and delayed germination responses follow early dry season fires, relative to late season fires in savanna (Williams et al., 2005). The **Endangered** *Rutidosis leptorhynchoides* (Button Wrinklewort) flowers in late spring and summer (Table 2). Therefore, spring and early summer fires not only prevent the production of seed in that year, but no recruitment takes place, reducing the likelihood of population persistence (Morgan, 1995; 1997). In temperate regions, spring fires can expose emerging seedlings to summer desiccation, which may cause high mortality and limit population recovery, especially in winter-rainfall zones of Western and South Australia (Miller et al., 2019; 2021). See section 3.2.2.1 Fire-drought interactions (FDR) for similar threats related to inter-annual droughts. In **Endangered** Banksia Woodland of the Swan Coastal Plain (Table 2) communities of Mediterranean climate in southwest Australia, seedlings that germinated after autumn fires were likely to survive through the following summer, but no seedlings that germinated after spring fires survived, threatening the structure, diversity and function of the Ecological Community (Miller et al., 2021; Roche et al., 1998).

Animal behaviour, reproduction, maturation and mobility are all strongly seasonal in many species, rendering their populations sensitive to fires that occur at particular times of year, most notably spring in temperate regions (Figure 1). In northern savannas, fires in the late dry season can reduce breeding success in populations breeding at that time, both through reduced nestling survivorship and fewer breeding attempts (Murphy et al., 2010). Three threatened mallee bird species, the **Endangered** *Stipiturus mallee* (Mallee Emu-wren), and the **Vulnerable** *Pachycephala rufogularis* (Red-lored Whistler) and *Psophodes nigrogularis leucogaster* (Western Whipbird) (Table 2), all build nests close to the ground during late winter and through spring (Department of Environment Land Water and Planning, 2016). Fires occurring during these periods will destroy nests and directly kill the eggs and nestlings, with impacts on population viability that depend on the longevity and fecundity of breeding individuals.

3.1.3 Fire severity (High severity, HSF; Low severity, LSF)

The causes of direct mortality and tissue damage by fire relate to lethal heat or asphyxiation by smoke. High severity fires are associated with greater exposure to lethal heat pulses and, hence higher mortality in both plants and animals (Chia et al., 2015a; 2015b) and fungi, while rapid rates of spread (related to high fire intensity, i.e. energy output) can limit the success of avoidance behaviour in animals (Nimmo et al., 2021). Lethal effects of smoke may extend beyond the burnt area. For example, bushfire smoke caused mortality in a captive population of the **Endangered** *Pseudomys fumeus* (Smoky Mouse) >20 km from a fire (Peters et al., 2021). Bushfire smoke has also been shown to affect wildlife behaviour in other parts of the world (Álvarez-Ruiz et al., 2021) and could be an important cue to behaviours that reduce mortality during fire events (Nimmo et al., 2021).

Extremes of fire severity above and below ground can pose a range of direct threats to plants, animals, fungi and ecological communities (Whelan, 1995). Below ground, heat transfer depends

primarily upon the mass of ground fuel consumed (Tunstall et al., 1976) and soil depth. Soil temperatures during fire have little relationship to above-ground fire behaviour (Cheney et al., 2012; Zylstra et al., 2016), or above-ground fire severity (Bradstock & Auld, 1995; Hartford & Frandsen, 1992; Penman & Towerton, 2008; Raison et al., 1986; Stoof et al., 2013).

As soil provides strong thermal insulation, many plants and animals exploit below-ground refuges from lethal temperatures. Some plants have buried regenerative organs (tubers, rhizomes, and so forth) or soil seed banks (Keith, 2012), while many invertebrates have subterranean life stages (Friend, 1996) and some vertebrates shelter in burrows (Whelan, 1995). High soil heating (Figure 1, mechanism HSF, high severity fires below ground) may pose a threat to some of these species because the survival rates of underground organs and life stages, and the safety of underground refuges, decline as soil heating increases (e.g., Bowman et al., 2009). However, soil heating diminishes rapidly with depth, so the impact of high fire severity below ground depends on the depth distribution of the organisms (Auld & Bradstock, 1996) and can be highly localised due to the patchy distribution of surface fuels consumed during the fire. Similarly to plants and animals, fungi species that are deep underground, or have heat-resistant spores, are likely much less sensitive, although fungi species associated with aerial parts of plants or with invertebrates are likely killed by severe fire (McMullan-Fisher et al., 2011).

The failure of fires to raise soil temperatures sufficiently (Figure 1, mechanism LSF, low severity fires) may also pose a threat to some plant species. For example, plant species with physical seed dormancy and some with physiological seed dormancy, if fire generates soil temperatures that are too low to alleviate seed dormancy (Auld & O'Connell, 1991). Recurrence of fires that heat soils insufficiently to initiate germination may cause population declines through attrition of seed banks (Regan et al., 2003). Many listed species from a range of plant families have such temperature-sensitive dormancy and germination, including the **Critically Endangered** *Grevillea caleyi* (Caley's Grevillea) (Table 2) and *Pultenaea* sp. *Genowlan Point* (NSW 417813) (Genowlan Point Pultenaea), **Endangered** *Pomaderris cotoneaster* (Cotoneaster Pomaderris), **Vulnerable** *Acacia bynoeana* (Bynoe's Wattle), *Darwinia biflora* and *Pultenaea glabra* (Smooth Bush-pea) (Auld & O'Connell 1991; Auld & Ooi 2009; Regan et al., 2003). In Caley's Grevillea, for example, low severity fires kill the standing plants, but few of the soil-stored seeds germinate to replace them. This removes the source of seed production without initiating seedling recruitment. Conversely, although high severity fires kill the standing plants, they generate abundant recruitment, sometimes increasing the population (Regan et al., 2003). Loss of these species from ecological communities have implications for persistence of avian and mammalian pollinators (Whelan, 1995), as well as invertebrates with larval stages or adults that are dependent on foliage, seeds or wood (Moir et al., 2011) (see section 3.2.1.1 Fire effects on habitat suitability and facilitation interactions (FHF), Figure 1).

Above ground, survival of both plants and animals declines as fire severity increases (Chia et al., 2015a; 2015b). Survival of plants typically depends on regenerative organs insulated from lethal heat by protective tissues (Bradstock et al., 1994; Wesolowski et al., 2014), while survival of animals depends on mobility (fire evasion) or ability to avoid heat in local refuges such as tree hollows, rock outcrops or plant crowns (Abbott & Whitford, 2002; Mason et al., 2018; Wayne et al., 2006; Whelan, 1995). Species in which such traits are absent or poorly developed may be prone to lasting declines or local extinctions if exposed to high severity fires unless mortality is counter-balanced by rapid and abundant post-fire recruitment or recolonization (Whelan et

al., 2002). Rapid rates of fire spread may also cause high mortality, even in mobile animals, due to asphyxiation from smoke as well as heat shock, and by limiting directional escape routes.

Examples of listed species threatened directly by high fire severity include the **Critically Endangered** *Pezoporus flaviventris* (Western Ground Parrot, Kyloring), the **Vulnerable** *Petauroides volans* (Greater Glider) and the **Endangered** *Phascolarctos cinereus* (Koala (combined populations of Queensland, New South Wales and the Australian Capital Territory)) (Table 2). Koalas, for example, were much more likely to be lost from sites burnt in 2019-20 when the canopy was impacted by fire (Law et al., 2022; Phillips et al., 2021).

Fire of any severity may initiate transformational changes in certain ecological communities. Rainforest communities, such as **Critically Endangered** Lowland Rainforest of Subtropical Australia and **Endangered** Monsoon vine thickets on the coastal sand dunes of Dampier Peninsula are among the most susceptible to long-term structural change when burnt (Kooyman et al., 2020; Threatened Species Scientific Committee, 2013), but similar responses to basal scorching are evident in some woodlands such as those dominated by *Eucalyptus salubris* in southwestern Australia (Gosper et al., 2018). Although intense fire behaviour in rainforest is rare, small flames (low fire severity) may have large and lasting impacts on structure and composition due to the thin bark of the trees, the potential for girdling (Brando et al., 2012) and slow recovery times after topkill. These effects are likely to be amplified if fires recur (HFF, Figure 1). Potential consequences include canopy opening, long-term dominance by vines, compositional change due to changes in fauna habitats (FHF) and differential mortality of tree species, loss of seed and seedling banks, weed invasion (FCI) and positive flammability feedbacks. Large structural and functional changes may also occur in some temperate eucalypt forests. This can reduce habitat complexity and enhance positive feedbacks related to rates of fuel drying, making them prone to flammability feedbacks and landscape fire traps (Lindenmayer et al., 2011; Zylstra et al., 2022). Soil heating can increase hydrophobicity and erosion rates, as well as impacting fungal and bacterial diversity (Pérez-Izquierdo et al., 2021; Santín & Doerr, 2016). Increased incidence of high severity fires in desert landscapes, in some areas partly driven by invasive grasses (see section 3.2.1.4 Fire effects on competitive interactions (FCI)), is predicted to reduce the abundance of large trees, such as marble gums (*Eucalyptus gonglyocarpa*) and River Red gums (*Eucalyptus camaldulensis*), which will have substantial long-term impacts for multiple dependent species that rely on the trees for habitat, including **Vulnerable** Princess parrots (*Polytelis alexandrae*)(see section 3.2.1.1 Fire effects on habitat suitability and facilitation interactions (FHF)) (Pavey et al., 2014; Schlesinger & Westerhuis, 2021).

High severity fires that consume all ground cover, particularly in riparian areas, may expose the surface to high risks of erosion if intense post-fire rainfall or wind events occur before re-establishment of groundcover vegetation. Loss of surface soil may reduce habitat suitability for terrestrial biota, but the consequential effects of sedimentation and ash inflows may have major effects on freshwater and coastal marine ecosystems and their associated species well beyond the immediate footprint of burnt areas (FHY, see section 3.2.2.2).

3.1.4   Fire type (Peat fire, PF)

Peat fires (Figure 1, mechanism PF), although much less common than surface or crown fires in Australia, pose unique and transformational threats to peat-associated species and ecological communities of cool temperate climates (French et al., 2016; Keith et al., 2020b). Peat

combustion depends on substrate dryness (Prior et al., 2020) and is unrelated to surface fire intensity (i.e. energy output) or surface fire severity (i.e. above-ground biomass consumption). Major substrate fires are associated with prolonged pre-fire droughts, although peat soils may be dry enough at some point in late summer of most years to support a degree of peat consumption if ignited (see Prior et al., 2020). Peat fires may smoulder for weeks or even months after ignition, causing high mortality of below-ground plant organs, seed banks, fungi and subterranean invertebrates, although slow rates of spread allow time for evasion by mobile animals (e.g., Corbett, 2010; Keith et al., 2020b). The consumption of peat by fire typically alters soil hydrology and chemistry, reduces soil water retention and also increases the erosion potential of residual mineral soils.

Ecological impacts of peat fires tend to be long-lasting, as they result in oxidised hydrophobic substrates that are unsuitable for many hydrophilic plants and subterranean animals. Threats posed by peat fires (PF, Figure 1) are greatest for species and ecological communities that are restricted to peat-accumulating ecosystems. Temperate examples include **Endangered** Alpine Sphagnum bogs and associated fens (Good et al., 2010), Temperate Highland Peat Swamps on Sandstone, and Coastal Upland Swamps in the Sydney Basin Bioregion (Keith et al., 2020b; In review), and species with limited mobility such as the **Critically Endangered** *Pseudophryne corroboree* and *Pseudophryne pengilleyi* (Corroboree frogs) (Table 2), which breed in saturated sphagnum moss (Hunter et al., 2009; NSW Office of Environment and Heritage, 2012).

3.1.5 Fire regime spatial patterns (FSP)

The core components of fire regimes (frequency, severity, season and type) vary spatially across landscapes, forming patterns or 'fire mosaics', including mosaics of antecedent fires (Bradstock et al., 2005). Fire mosaics may be viewed as configurations of patches or elements with varying combinations of size, dispersion, severity, season, frequency and post-fire age (Table 1). Certain fire mosaics may threaten biodiversity (Martin & Sapsis, 1992) if they are unsuitable for habitation by particular species (FSP, Figure 1). Conversely, changes in fire mosaics may initiate recovery of threatened species such as the **Endangered** *Erythrura gouldiae* (Gouldian finch) (Legge et al., 2015), indicating its sensitivity to spatial patterns in fire regimes. Fire mosaics may also promote species coexistence, for example, as post-fire abundances of desert species, such as spiders (Langlands et al., 2012) and reptiles (Letnic et al., 2004; Pianka & Goodyear, 2012) vary out of phase with one another. The effects and mechanisms of the postulated threats are complex, mostly indirect and still poorly understood, but several different hypotheses relating to variation in mosaic composition (the number, type and extent of fire elements) and configuration (the arrangement of fire elements) are under investigation (Griffiths et al., 2015; Kelly et al., 2017; Lawes et al., 2015; Legge et al., 2019).

First, some animals are postulated to use resources that are available only from patches in different post-fire states and hence they rely on movement through fine-scale mosaics of different post-fire ages (landscape supplementation, Nimmo et al., 2018), which may be disrupted by large fires. Correlative evidence suggests that **Endangered** *Lagorchestes hirsutus* Central Australian subspecies (Mala, Rufous Hare-Wallaby (Central Australia)) (Table 2) and the unlisted *Varanus gouldii* (Gould's Goanna or Sand Monitor) feed in recently burnt patches and shelter in longer unburnt patches (Bliege Bird et al., 2018; Lundie-Jenkins, 1993). Birds such as the **Vulnerable** *Geophaps smithii* (Partridge Pigeon (eastern)) and *Geophaps smithii blaauwi* (Partridge Pigeon (western)) decline when fires exceed their home range size, especially during the breeding season, as they require both open ground to forage, and dense vegetation to nest

(Fraser et al., 2003). Where fire mosaics change such that fire patch size exceeds movement capability and perceptual ranges of the animals, this could lead to resource deprivation and population decline, depending on the degree of reliance on resources limited to the patch types (Nimmo et al., 2018).

Second and conversely, animals dependent on coarse-scale mosaics of mature post-fire age may be sensitive to patchy fires that disrupt habitat continuity. The **Endangered** *Manorina melanotis* (Black-eared Miner) is positively associated with a low diversity of age classes, specifically large patches of mallee unburnt for many decades (Taylor et al., 2013). Small patches of long-unburnt mallee are unoccupied and evidently unsuitable for habitation, although the evidence is correlative only. A positive association with the amount (or extent) of long-unburnt vegetation in a landscape has been documented for several bird, mammal and reptile and species that co-occur with the Black-eared Miner (Farnsworth et al., 2014; Kelly et al., 2012; Nimmo et al., 2013; Taylor et al.2013) (Table 2).

Third, movements and foraging efficiency of predators and herbivores may be enhanced in small or patchy fires, which result in elevated mortality or suppression of growth and reproduction of prey and palatable plant species in the post-fire environment or during fire events (Hradsky, 2020; Leigh & Holgate, 1979). This mechanism may involve introduced or native predators and herbivores (see section 3.2.1.2 Fire effects on predator-prey interactions (FPI) and section 3.2.1.3 Fire effects on herbivore-plant interactions (FHI), Figure 1). Stochastic population models for the **Critically Endangered** Caley's Grevillea (Table 2) based on empirical demographic data, indicate that small patchy fires increase the impact of granivores on seed bank accumulation after fire, increasing the extinction risk of populations to high frequency fire (see section 3.1.1 Fire frequency (High frequency, HFF; Low frequency, LFF), Figure 1) relative to larger fires, which are associated with lower levels of seed predation by granivores (Regan et al., 2003).

Although all three mechanisms are plausible and supported by anecdotal observations or indirect inferences, the empirical evidence linking mosaic fire patterns to measures of biodiversity in Australia is so far limited to a small number of studies (Bleige Bird et al., 2018; Parr & Andersen, 2006; Pastro et al., 2011). For example, although a positive relationship between 'pyrodiversity' and biodiversity has been suggested, several studies have found weak or no relationships between fire patch diversity and biodiversity (e.g., Griffiths et al., 2015; Nimmo et al., 2013; Taylor et al., 2012), and more work is needed to resolve mechanisms, dependencies, and variation in the drivers across Australia.

3.2 Threats operating indirectly via ecological interactions and other processes

A range of biotic interactions (facilitation, predation, competition) and abiotic processes (e.g., hydrological or edaphic functions) may be disrupted by particular fire regimes (Figure 1). Fire regimes may also amplify threats related to land use, and fire management operations that manipulate structural or chemical properties of ecosystems. Finally, feedbacks between anthropogenic climate change, fire regimes, and biota can also produce complex interactive threats to biodiversity. For some species and ecological communities, complex inter-dependencies between multiple biotic processes that interact with fire regimes make threats difficult to diagnose and predict. In northeast Australia, the **Endangered** Alwal, Golden-

shouldered Parrot is at risk from an interaction between fire regimes and grazing that promotes shrubs over the grass species that provide the animal's food source (Bateman & Johnson, 2011; Reside et al., 2016). The **Endangered** *Trisyntopa scatophaga* (Antbed Parrot Moth) is in-turn entirely dependent upon the nests of the parrot, so that the loss of one species would result in the loss of the other (Threatened Species Scientific Committee, 2016a). Developing knowledge on interactive threats indicates that they may be as important, or sometimes more important, in mediating persistence or extinction outcomes for biodiversity than threats that stem directly from the fire regime itself.

3.2.1   Fire effects on biotic interactions (FHF, FPI, FHI, FCI, FDI)

Ecosystem function encompasses many positive and negative interactions among component species. Recurring fires may regulate these interactions, disrupting or amplifying them through several different mechanisms that either maintain coexistence or cause extinctions.

3.2.1.1 Fire effects on habitat suitability and facilitation interactions (FHF)

Some fire regimes may disrupt facilitation or mutualistic interactions (FHF, Figure 1) so that fire-induced decline or loss of one or more species initiates decline in the viability of dependent species. The most common expression of this process is where one group of species depends on habitat resources (e.g., shelter, food, breeding sites) provided by another group: particular fire regimes may reduce habitat suitability of the dependent group by causing declines or altering population structure in the group providing habitat (Andersen, 2021).

Many of these facilitation interactions involve dependencies on populations of structural or functional dominants or keystone biota such as forest eucalypts (Lindenmayer et al., 2013), groundlayer plants (e.g., sphagnum moss, Hunter et al., 2009), hummock grasses (Haslem et al., 2011), savanna trees and grasses (Andersen, 2021), and fossorial mammals (Lamont, 1995). For example, yellow-bellied gliders (*Petaurus australis*) appear sensitive to the fine-scale structure under the canopy of potential feeding trees, and this could make them susceptible to changes in fire regimes, and interactions between fire and other potential threats such as logging (Heise-Pavlov et al., 2018). In another example, frequent fire has caused severe declines in populations of **Endangered** *Banksia montana*; with that loss, the **Critically Endangered** Banksia montana mealybug (*Pseudococcus markharveyi*), has also declined, possibly to extinction, because it depended entirely on that single host plant for survival (Moir, 2021).

Declines in the dead remains of woody plants caused by high frequency fire (HFF) may disrupt facilitation interactions (FHF, Figure 1) by reducing the carrying capacity of habitats for dependent vertebrates, invertebrates, and fungi. This may occur because frequent fire increases the loss of woody debris and hollow-bearing trees relative to their rates of development, resulting in a net decline in availability of the resource. For example, prescribed burning reduced den sites for the unlisted *Antechinus flavipes* (Yellow-footed Antechinus) in large trees, dead trees and logs by 30 percent (Flanagan-Moodie et al., 2018). **Critically Endangered** *Gymnobelideus leadbeateri* (Leadbeater's possum) preferentially occupy hollows in standing dead trees, many of which originate from fire-caused mortality, but which undergo accelerated rates of collapse when burnt in high or low severity fires (Lindenmayer et al., 2021; 2013) (Table 2). More generally, reduced density and stature of Alpine ash and Mountain ash forests caused by high frequency fire (HFF), poses threats to a range of fauna dependent on tree-related resources such as hollows, large canopies, loose bark and woody debris (Burns et al., 2015). In other forest environments treated with low-severity prescribed burns, hollow-bearing trees

declined by up to 26 percent (Inions et al., 1989; Parnaby et al., 2010). These threats may be further amplified by logging (FFL) (Lindenmayer et al., 2012). In northern Australia, high frequency fire (HFF), especially of high severity fires, are related to declines in the availability of tree hollows on which many fauna species depend (Woolley et al., 2018).

Large diameter, decaying woody debris is important habitat, particularly for small reptiles in forests (Grove & Meggs, 2003; Manning et al., 2013), as well as bracket fungi (polypores) and invertebrates such as beetles (coleopterans). Threats to these facilitation interactions (Figure 1) depend on the frequency of high- and low-severity fires. High-severity fires (HSF), especially those with long residence times, consume or accelerate decay of woody debris, but may also generate new woody debris from living biomass, depending on forest structure, fuel moisture and prior fires. Whitford & McCaw (2019) found that the total volume of coarse woody debris in jarrah production forest was positively correlated with the frequency of prescribed fire and negatively correlated with the frequency of wildfires, while Bassett et al. (2015) found that infrequent, high severity fires increased coarse woody debris in eastern forests. In arid woodlands Gosper et al. (2019) found that large diameter woody debris increased in volume with time since fire.

Shrubs in forest understories, heathlands and arid shrublands provide food, shelter and breeding sites for a wide range of birds, mammals, reptiles and arthropods (e.g., Bradshaw & Bradshaw, 2017; Brooker & Rowley, 1991; Hunter et al., 2009; Moir et al., 2011). While high-severity fires (HSF) result in temporary loss and re-establishment of these resources, re-establishment of dependent species depends on conditions for recruitment or immigration (see section 3.2.2.1 Fire-drought interactions (FDR) and section 3.1.5 Fire regime spatial patterns (FSP)), and declines in shrub resources may be driven by high frequency fires (HFF) or out-of-season fires (OSF).

Fire-induced declines or losses of individual plant species from ecological communities have implications for persistence of avian and mammalian pollinators (Bradshaw & Bradshaw, 2017; Harris et al., 2014), as well as invertebrates with larval stages or adults dependent on foliage, seeds or wood (Moir et al., 2011; Moir, 2021). Similarly, orchid populations may exhibit indirect negative responses to fire if their mycorrhizal fungi are suppressed by smoke- and charcoal chemical products generated by fires (Jasinge et al., 2018a). Risks of co-extinction are likely to be greater for species with narrow host ranges (Moir et al., 2011; Moir, 2021).

3.2.1.2 Fire effects on predator-prey interactions (FPI)

An important group of threats involves interactions between fire regimes, predators and their prey (FPI, Figure 1). Fire alters movement behaviour and can increase foraging efficiency in a range of mobile vertebrate predators, including introduced foxes and cats (Hradsky, 2020). Sensory cues enable predators to converge on burnt areas during and after fire (Hovick et al., 2017; Nimmo et al., 2018), sometimes over long distances outside home ranges (McGregor et al., 2016). Increased predator densities, or reduced shelter (see section 3.2.1.1 Fire effects on habitat suitability and facilitation interactions (FHF)), can individually or together expose prey species to greater risks of mortality than either fire or predation alone (Davies et al., 2020; McDonald et al., 2016; McGregor et al., 2015). These interactive threats may be more severe under high severity and high frequency fire regimes, as they increase or prolong exposure to predators, respectively (Hradsky, 2020; Leahy et al., 2016; Stobo-Wilson et al., 2020; Stokeld et al., 2018). These threats may also be exacerbated through herbivore-plant interactions

(FHI, section 3.2.1.3) such as cattle grazing (Legge et al., 2011; 2019), which reduce the availability of shelter.

Fire regimes may also modify interactions among predators, contributing to complex networks of trophic interactions. For example, in extensive mallee ecosystems, there is evidence that dingo activity increased in burnt areas, triggering avoidance behaviour in foxes, although there was no evidence of predator release in small mammal prey (Geary et al., 2018).

Fire effects on predator-prey interactions influence a wide range of prey species, although ground-nesting birds and critical weight-range mammals are likely to be among the most susceptible. These include the **Endangered** *Bettongia penicillata ogilbyi* (Woylie) (Christensen, 1980), the **Endangered** *Bettongia tropica* (Northern Bettong) (Whitehead et al., 2018), *Isoodon obesulus* (Southern Brown Bandicoot (south-eastern)) (Robley et al., 2014), and the **Critically Endangered** (Western Ground Parrot, Kyloring), (Comer et al., 2020) (Table 2). There is also evidence of enhanced post-fire predation of **Vulnerable** Greater Glider species by native Powerful owls (Kavanagh, 1992).

3.2.1.3 Fire effects on herbivore-plant interactions (FHI)
Predators of plants – herbivores and granivores – also interact with fire in ways that amplify threats to the plants (FHI, Figure 1). These animals may be attracted to burnt areas by post-fire regrowth, which is apparently more palatable than mature vegetation, or by an abundance of seeds released from canopy seed banks. Further, there is some evidence that physical and chemical defences of plants can be weaker in regrowth during the immediate post-fire period, and less developed in seedlings, which in many species tend to recruit more in the post-fire period than other times during fire intervals (Keith, 2012). This threat (FHI) pertains to both herbivory and trampling of regenerating plants and soils. Leigh and Holgate (1979), and Leigh et al. (1991) found empirical evidence of strong fire-herbivore interactions suppressing survival, growth and reproduction in a range of forest and woodland understorey plants in temperate and subalpine climates. Giljohann et al. (2017) found strong empirical evidence of similar adverse effects of herbivores on survival of seedlings and resprouts of *Triodia scariosa* (spinifex) during the first post-fire year. *Triodia* hummocks are a foundation resource for arid ecosystems, providing food and shelter to a diverse assemblage of vertebrate and invertebrate ground fauna (Haslem et al., 2011). Declines and local extinctions of *Triodia* are likely to have cascading effects on ecosystem function and diversity (FHF, Figure 1).

The species and ecological communities most likely to be affected by Fire-herbivore interactions include slow-growing palatable shrubs, forbs and flowering orchids that co-occur with grasses within livestock production landscapes, or areas with high densities of feral or native herbivores. Examples include the **Endangered** *Pterostylis basaltica* (Basalt Greenhood) (Ingeme & Backhouse, 1999) and **Critically Endangered** White Box-Yellow Box-Blakely's Red Gum Grassy Woodland and Derived Native Grassland (Lunt et al., 2007). Feedbacks exist between fire and herbivore-plant interactions, whereby animal diversity and abundance can affect fire regimes (Foster et al., 2020) by altering flammability through vegetation biomass, composition, and structure.

3.2.1.4 Fire effects on competitive interactions (FCI)
Fire-competition interactions may pose a threat when fire promotes the establishment of high-density populations of invasive plants that exclude inferior native competitors, or transform the structure and composition of native plant communities, thereby reducing habitat suitability for

some native animals (FCI, Figure 1). Fires can accelerate invasion processes by creating gaps for the entry of superior competitors, especially in sites predisposed by fragmentation (see section 3.2.1.1 Fire effects on habitat suitability and facilitation interactions (FHF), Figure 1) or eutrophication. Examples include post-fire invasions by exotic grasses in **Endangered** Banksia Woodland of the Swan Coastal Plain (Milberg & Lamont, 1995), **Endangered** Brigalow (*Acacia harpophylla* dominant and co-dominant) (Butler & Fairfax, 2003) and **Critically Endangered** Cumberland Plain Shale Woodlands and Shale-Gravel Transition Forest (Hill & French, 2003) (Table 2) – although intermediate fire intervals can also encourage weed invasion in this community (Watson et al., 2009). Fire can also interrupt nutrient cycling in phosphorus-limited, diverse temperate ecosystems such as the **Endangered** Banksia Woodland of the Swan Coastal Plain, enhancing fire-weed feedback loops (Fisher et al., 2009).

Fire regimes also mediate competition between co-occurring native species, promoting coexistence by interrupting the exclusion of inferior competitors. Conversely, low fire frequencies may allow time for elimination of inferior competitors. These fire-competition interactions are likely to be a threat in ecosystems that are capable of developing high-density populations of dominant plants, especially where limitations on dispersal and seedbank longevity limit recovery potential of inferior competitors (Morgan, 1995). Examples include exclusion of shade-intolerant sclerophyll plant taxa by later successional species (Baker et al., 2020), exclusion of small shrubs by larger shrubs in heathlands (Keith et al., 2007), and exclusion of interstitial forbs by large tussocks in productive temperate grasslands (Lunt & Morgan, 2002; Morgan, 1997). The **Endangered** Button Wrinklewort has short-lived standing plants and seedbanks and exhibits gap-dependent recruitment, requiring disturbances such as recurrent fire to provide the gaps for seedling recruitment (Morgan, 1995). Low frequency fire (LFF, Figure 1) may predispose it to competitive elimination by grass tussocks (FCI, Figure 1) (Table 2) unless there are other mechanisms of gap creation enabling recruitment. Ecological communities threatened by these fire-mediated autogenic processes include **Endangered** Coastal Upland Swamps in the Sydney Basin Bioregion (Keith et al., 2007) and **Critically Endangered** Natural Temperate Grassland of the Victorian Volcanic Plain (Morgan, 1998; Williams et al., 2006) (Table 2).

3.2.1.5 Fire effects on disease interactions (FDI)

Fire regimes interact with pathogens to produce elevated rates of mortality and reduced rates of growth and reproduction (FDI, Figure 1). Most evidence comes from reports of increased prevalence of symptoms of plant diseases in the early post-fire years, notably *Phytophthora cinnamomi* (root rot) (Moore et al., 2014) and *Austropuccinia psidii* (Myrtle rust) (Pegg et al., 2020). Regan et al. (2011) found that mortality of *Xanthorrhoea resinosa* peaked in the first and second years of the fire cycle, with *P. cinnamomi* isolated from roots of dying plants. Consequently, they found that the demographic impacts of fire-disease interactions could be mitigated by reducing fire frequency.

How fires amplify disease impacts is poorly understood, and probably varies between pathogens. Potential mechanisms include enhanced pathogen dispersal by water or wind in post-fire conditions, increased sporulation as a result of fire-released nutrients, elevated soil moisture levels, increased soil warming, increased vulnerability to infection and/or necrosis of young post-fire regrowth or seedling tissues, and additive physiological stresses during re-establishment in the post-fire environment. Fires may also amplify impacts of diseases associated with trophic dysfunction, such as rural dieback (Landsberg, 1988) and

Manorina melanophrys (Bell-miner) associated dieback (Florence, 2005; Wardell-Johnson & Stone, 2006), as disease is expected to weaken capacity for recovery of trees after fire. Analogous mechanisms for interactions between fire and animal diseases, including changed movement patterns of disease vectors, are mostly yet to be investigated (though see Ecke et al., 2019). Examples of threatened species and ecological communities threatened by fire-disease interactions include **Endangered** *Epacris barbata* (Bearded Heath) (Keith, 2004), **Endangered** Coastal Swamp Sclerophyll Forest of New South Wales and South East Queensland (Pegg et al., 2020) and **Endangered** Eastern Stirling Range Montane Heath and Thicket (Barrett & Yates, 2015) (Table 2).

3.2.1.6 Interaction cascades

In addition to the main groups of biotic interactions influenced by fire regimes, fires may also mediate higher order interactions (interactions between interactions). In such networks of dependencies, perturbation of some elements by particular fire regimes may generate cascades of declines or extirpations, and shifts in the composition, structure and function of ecological communities and ecosystems. The dependencies are complex and too little evidence exists to draw generalisations about fire-mediated interaction cascades at this time, however, examples of these relationships are beginning to emerge. For example, Legge et al. (2019) found that fire-predator interactions (FPI, Figure 1) may be mitigated by reducing fire prevalence in the landscape, but this effect was offset by livestock grazing which suppress vegetation cover, presumably with long-lasting effects where high densities of livestock are present in recently burnt areas (FHI, Figure 1). Many other cascades, such as fire-mediated exclusion of a facilitation species seem plausible but are yet to be investigated.

3.2.2 Interactions between fire regimes and abiotic processes (FDR, FHY, FFC, FFL, FSD, FSR)

Ecological responses to fire vary greatly depending on the availability of resources and on other aspects of the environment that regulate or impede resource uptake. The best known of these interactions that threaten biodiversity are related to water availability, quality and its movement through landscapes and soils (Gallagher, 2020; Gallagher et al., 2021; Keith et al., 2020b; In review). Understanding of short-term and local effects of fire on *in situ* soil chemistry, structure and biota is improving, and some evidence is beginning to emerge that these processes could threaten biodiversity.

3.2.2.1 Fire-drought interactions (FDR)

Water is a key biological resource, and often the most limiting one that influences the health and condition of individuals prior to a fire event (through accumulation of fat or starch reserves) and their ability to recover and reproduce in the post-fire environment. The timing of fire in relation to inter-annual droughts can therefore have profound effects on population outcomes and extinction risks (Choat et al., 2018; Hale et al., 2016). Threats related to fire-drought interactions (FDR, Figure 1) potentially affect a wide range of species and ecological communities from relatively well-watered tropical forest, wetlands and alpine systems to deserts where extreme water deficits are the norm. When fires are preceded by wetter than average years, animals build fat reserves that may equip them for a period of resource deprivation in the post-fire environment. Conversely, pre-fire drought may predispose populations to high mortality in the post-fire environment, other things being equal, because their reduced physiological condition cannot sustain them through a further reduction in resource availability. In aquatic systems, pre-fire drought can cause marked contractions in the range and occurrence of fish and other

aquatic fauna, making them more vulnerable to impacts from fire. Pre-fire drought may, on the other hand, reduce pre-fire populations of browsers. In plants, fire-drought interactions influence the risk of hydraulic failure and mortality (Choat et al., 2018), with potentially long-term impacts on ecosystems and habitats.

Interactions between fire and post-fire droughts are best understood for plants. Obligate-seeding plants are particularly exposed to this threat because many rely primarily on post-fire seedling recruitment for population persistence, and seedlings have rudimentary root systems prone to desiccation in their establishment phase. Burgman and Lamont (1992) found that extinction risk of the **Endangered** *Banksia cuneata* (Matchstick Banksia) (Table 2) was more sensitive to recruitment failure due to post-fire drought than fire frequency when intervals between successive fires were greater than five years. Their population model showed that extinction risks were lowest when fire intervals approached the longevity of standing plants, which was a much longer fire interval than when seed bank size reached a peak. Resprouting woody plants are also susceptible to elevated mortality in post-fire droughts because their regrowth tissues are more susceptible to xylem embolism than mature tissues (Pratt et al., 2014). While this has implications for persistence of threatened fire-prone forest and woodland communities under climate change, investigations are at an early stage (Choat et al., 2018; de Kauwe et al., 2020).

Apparent synergistic negative effects of fire and drought have been suggested for a range of mammal and bird species. In coastal heath and desert, respectively, Recher et al. (2009) and Dickman et al. (2014) both noted a decline in abundance and richness of small mammals that lasted several years after fire until substantial rains recharged the system, while Bennison et al. (2018) found that post-fire exploitation of burnt areas by rodents and small marsupials was curtailed by post-fire drought. Evidence of elevated extinction risks for such species is challenged by limitations on detectability and high population variability, but the **Critically Endangered** *Zyzomys pedunculatus* (Central rock-rat) (Table 2) apparently disappeared from long term trapping sites in response to extended drought, with fire possibly also an explanatory factor (Dickman et al., 2014; Nano et al., 2019).

3.2.2.2 Fire-hydrological interactions (FHY)

Fires liberate a large quantity of nutrients and other compounds sequestered in standing vegetation. While some is volatilised, some is leached through the soil profile and some is recaptured by regenerating plants and animals, a portion of the nutrient capital is washed by overland flow after rain into streams and other water bodies. Severe fires that consume the riparian vegetation, when followed by intense or prolonged precipitation events, may result in a large pulse or 'slug' of nutrient and sediment input into water bodies (Alexandra & Finlayson, 2020; Gomez Isaza et al., 2022; Nyman et al., 2019). These slugs can pose threats to a range of aquatic biota (FHY, Figure 1). For example, they may cause significant mortality and habitat transformation, elevating extinction risk for aquatic biota and associated communities, particularly fish, amphibians and macro-invertebrates that are dependent on low-nutrient, high-oxygen or low-turbidity aquatic habitats (Cramp et al., 2021; Gomez Isaza et al., 2022; Lyon & O'Conner, 2008; Silva et al., 2020). The severity of soil heating (HSF, LSF, section 3.1.3) affects the chemistry and scale of the slug (Cawson et al., 2016; Santín & Doerr, 2016). There are few studies on specific effects of slugs on aquatic life (e.g., Douglas et al., 2003; Lyon & O'Conner, 2008), but negative impacts of bushfire ash, including reduced physiological function and elevated mortality, have been demonstrated in laboratory experiments (Cramp et al., 2021;

Gonino et al., 2019) and health effects on humans provide relevant insights (Abraham et al., 2017). Hines et al. (2020) suggested postfire sedimentation, accumulation of debris and changes in water chemistry had impacts on the population of the **Endangered** *Taudactylus eungellensis* (Eungenella day frog). Fire severity is likely to influence the magnitude of impacts, while fire frequency may limit recovery processes and mediate declines.

Dense, fast-growing regrowth can consume large quantities of soil water, reducing surface flow and seepage into streams during the post-fire years of regrowth. This effect is more pronounced in forests dominated by tall obligate-seeding tree species (Buckley et al., 2012; Kuczera, 1987) than in forests of resprouting trees (Gharun et al., 2013). The potential declines in aquatic biota that result from prolonged reduction in stream flow and recovery of forests are yet to be studied.

Terrestrial ecological communities largely confined to steep slopes or unconsolidated substrates may also be prone to large erosion or sedimentation events associated with intense post-fire rainfall events and extreme overland flows, disrupting or inhibiting re-establishment of post-fire vegetation. All fires increase the likelihood of erosion because they remove surface protection and modify soil hydrophobicity. The magnitude of erosion depends on intensity of post-fire rainfall events, fire severity and patchiness, as well as site specific components such as slope and soil properties (Renard et al., 1991; Teng et al., 2016; Yang, 2020). Fire effects may be exacerbated when clearing, logging or grazing activity disrupts soil stabilisers such as subsurface root strata and soil crusts (Eldridge & Greene, 1994).

Interactions between fire and hydrology may also occur when prior alterations to hydrology erode ecosystem resilience and predispose species or ecosystems to amplified impacts from subsequent fire. There is strong evidence of such effects in **Endangered** Temperate Highland Peat Swamps on Sandstone affected by underground longwall coal mining (Table 2; Keith et al., 2020b; In review). Extraction of the coal seam and subsequent collapse of overburden rock into the void reduces the capacity of the swamps to retain moisture because surface cracking and warping, increases substrate permeability beneath the swamps and alters surface flows and seepage patterns. When burnt, the relatively dry vegetation and soils enable peat fires (PF, Figure 1), with associated threats described in section 3.1.4.

The effects of fire regimes on aquatic biota and ecosystems may extend well beyond the area burnt in fire events, downstream along watercourse and to the ocean. While fires may directly affect intertidal communities such as mangroves and saltmarsh, bushfires in catchments can result in mortality in fish and other marine life in estuaries and on beaches (Ley & Chester, 2021; Manjalay, 2020; Silva et al., 2020). Such impacts are associated with altered water quality, either through fluvial outflows or through aerosol fallout. Effects of changes in water quality and chemistry in shallow coastal waters have been reported for soldier crabs, sea urchins, soft sponges, turban snails, and some other marine invertebrates, while invasive species may recolonise affected areas faster than native species (Abyss Project). Aerosol effects of major fire events may generate biogeochemical responses in remote marine waters (Li et al., 2021). Although these studies suggest potential for threats to marine biodiversity from fire-hydrological interactions (FHY), particularly in response to high severity fires (HSF), more research is needed to improve understanding of the mechanisms, sensitivities and recovery potential.

3.2.2.3 Fire-fragmentation effects through vegetation clearing (FFC)

Fragmentation of terrestrial ecosystems by clearing of native vegetation and replacement by agricultural or urban/industrial systems reduces the complexity and diversity of natural landscapes, and transforms landscape structure. Fragmented landscapes are characterised by patches of native vegetation, modified to varying degrees, within a transformed matrix that is unsuitable for much of the original flora and fauna (Fahrig, 2003; Haddad et al., 2015). Landscape fragmentation by clearing reduces biodiversity, and also alters landscape fire regimes, with major implications for persistence of residual plants and animals through fire regimes (Driscoll et al., 2021). Land clearing has mixed feedbacks on fire regimes, depending on the vegetation that replaces the original ecosystems. Some studies suggest that land clearing has been responsible for rainfall decline, leading to not only a drier landscape, but also a more flammable forest structure (Andrich & Imberger, 2013; Ruthrof et al., 2016).

Clearing converts larger areas of native vegetation to fragments within a matrix of pastures, crops, plantations or urban/industrial infrastructure. The transformed ecosystems typically have altered fuel structure and volumes, and altered microclimates. These factors have major implications for fire spread, depending on fuels in the altered matrix surrounding remnant patches. In some rural settings, for example, landscape fragmentation can result in the long-term exclusion of fire from isolated pockets of vegetation (Gosper et al., 2013). Conversely, in peri-urban and some other settings, the interfaces may be exposed to increased frequency of human ignitions, depending on adjacent land use, social systems and human activity.

The interaction between fragmentation and fire regimes affects extinction risks in several ways (FFC, Figure 1). First, fragmentation reduces the area and quality of certain types of habitat in the landscape (Haddad et al., 2015), reducing the carrying capacity for recovery and persistence of species populations under recurring fires. Second, fragmentation inhibits movement of some species, limiting opportunities for post-fire recolonization by species that are unable to persist through or take refuge in situ during the fire event (Nimmo et al., 2018). Third, fragmentation establishes extinction debts (Tilman et al., 1994), whereby inviable populations of plants and animals persist for some time after fragmentation occurs. Extinctions may be forestalled for extended periods, especially in longer lived species, or those where reproduction partially compensates mortality. By increasing mortality and population turnover, fires may accelerate lagged rates of species loss. A similar process occurs at the scale of ecosystems, where the loss of important ecosystem components may set collapse in motion, but the loss may be undetected for a prolonged period (Lindenmayer & Sato, 2018). For example, fragmentation associated with cropping isolated populations of the **Endangered** *Verticordia fimbrilepis* subsp. *fimbrilepis* (Shy Featherflower), then prevented fires from spreading into the remnants. It appears that, given its low numbers and isolation, the species is now dependent on intermittent fires for replenishment of its relatively short-lived seed bank (Gosper et al., 2013; Yates & Ladd, 2010). Finally, fragmentation may activate other mechanisms of threat indirectly by altering fire regimes, in turn promoting invasion of competitors (e.g., weeds) or predators (Milberg & Lamont, 1995).

3.2.2.4 Fire-fragmentation interactions and other effects associated with logging/timber harvesting (FFL)

Timber harvesting involves the removal of trees from forest ecosystems, thereby altering their structure and fragmenting the landscape distribution of forest age classes. Unlike land clearing and land use conversion, it is followed by active or passive tree regeneration and is likely to be repeated over time to maintain supply of wood products. This results in a transformation of

habitat, microclimate and bushfire fuels that differs both from fragmented rural and urban landscapes and from unharvested forest. Removal of large trees and pulsed production or rearrangement of woody debris selectively reduces habitat suitability and carrying capacity for species dependent on those components of forest ecosystems. These effects vary with silvicultural systems (Lindenmayer et al., 2006; Schutz, 2002) and the changes can interact cumulatively with planned or unplanned fires at different times in the regenerative process, elevating threats to affected animals and plants through additional mortality and by prolonging reductions in carrying capacity (FFL, Figure 1). There also is evidence that rapid and dense regrowth after logging, together with greater exposure and drying of ground fuels, increases flammability of regrowth forest for a number of decades, potentially resulting in 'landscape trap' feedbacks (Furlaud et al., 2021; Lindenmayer et al., 2011; 2021; Taylor et al., 2021b; 2014). Despite the evidence for positive fire feedbacks associated with harvesting, extreme fire weather and low fuel moisture remain the major drivers of large and severe fires (Bowman et al., 2021a; Keenan et al., 2021).

Mechanisms of threat and their effects depend on whether timber harvesting precedes fire (green logging) or vice versa (salvage logging). Forests regenerating after green logging may undergo major long-term structural transformation if burnt during the early decades of the regenerative phase. Risks of ecosystem collapse from these processes appears to be greatest in rainforests that have legacies of past logging activity, such as the **Critically Endangered** Lowland Rainforest of Subtropical Australia (Threatened Species Scientific Committee, 2011) and wet eucalypt forests (Burns et al., 2015), as these are structurally complex ecosystems in which fire spread is most likely to be limited by moist microclimates.

Salvage logging removes scarce remaining post-fire habitat resources including large live trees, dead stags and wind-thrown trees, fallen timber and tree hollows, causing marked decline in cavity-dependent and saproxylic (dead wood dependent) organisms (Thorn et al., 2018). Salvage logging also has indirect effects (Lindenmayer & Noss, 2006) including damage to non-target species such as tree ferns (Bowd et al., 2018) compaction and erosion of exposed soils, and long-term (>80 years) nutrient depletion (Bowd et al., 2019). The species most sensitive to interactions between fire and both types of logging include birds and mammals that depend on live and standing dead large trees (Lindenmayer et al., 2018), such as the **Critically Endangered** Leadbeater's possum and the **Vulnerable** Greater Glider (Table 2).

3.2.2.5 Fire suppression disturbance (FSD)

Physical disturbances associated with firefighting operations and post-fire 'mop up' (FSD, Figure 1) include construction of roads and fire control lines, earthworks, removal of trees and expansion of burnt areas when backburning is carried out to contain wildfires (Driscoll et al., 2010). Some firefighting operations involve dousing fires with seawater or water removed from streams, lakes or wetlands. These activities are based largely on reactive decisions in emergency situations, posing challenges for mitigating impacts on biodiversity through improved planning and communication. Although impacts are localised, they pose disproportionate threats to highly restricted species and may have cumulative impacts over multiple fire events. They may also exacerbate other mechanisms of threat. For example, earthworks and fire lines may promote weed invasion (FCI, Figure 1) and predator activity (FPI, Figure 1) in the post-fire environment. The most susceptible species to fire suppression disturbances are plants and invertebrates with very restricted distributions and limited

dispersal abilities, particularly those found on ridges, wildland-urban interfaces and other potential control lines.

3.2.2.6 Fire retardants (FSR)

Firefighting operations may involve chemical, as well as physical disturbances to the environment. Rapid technological development of fire suppression methods is likely to intensify impacts on biodiversity over more extensive areas.

Aerial application of fire-retardant chemicals during fire suppression operations has become widespread in recent years, notably in the 2019-20 fire season. Likely increases in extent and frequency of such applications in future are indicated by acquisition of capital equipment and large air tankers (Commonwealth of Australia, 2020). These chemicals are broadcast ahead of the fire front in remote areas and on bushland interfaces with urban or rural infrastructure. Examples of fire-fighting chemicals applied by aircraft in Australia include Phos-Chek®, Blazetamerand and Thermo-Gel. Their main chemical constituents include ammonium sulphate, diammonium sulphate, and ammonium phosphate, with guar gum thickener, corrosion inhibitors and iron oxide pigments (NSW Rural Fire Service, undated).

Research on the ecological effects of fire-retardant chemicals in Australia is so far very limited (Leach, 2013). However, there is evidence of negative effects on plants and aquatic ecosystems (FSR, Figure 1). Although data on threatened species are lacking, Phos-Chek has been shown to promote weeds, and to reduce growth and increase mortality in sclerophyll plant genera such as *Allocasuarina*, *Banksia* and *Leptospermum* (e.g., Bell et al., 2005), which include multiple taxa listed as threatened under the EPBC Act. Inherently low levels of phosphorus and nitrogen in many Australian ecosystems, particularly fire-prone systems (Hopper, 2009; Lambers et al., 2010), are likely to make them and their component threatened species sensitive to fire retardants containing ammonium (hence nitrogen) and phosphates. Increased levels of these and other nutrients produce toxicity symptoms in sclerophyllous ecosystems and promote the invasion of introduced plants (Heddle & Specht, 1975; Lambers et al., 2013). Aquatic freshwater ecosystems are particularly sensitive to nitrogen and phosphorus enrichment (Bowmer, 2013), with algal blooms among the likely responses. Some of these effects may be cumulative, particularly those related to phosphorus. Effects on terrestrial animals are unknown and are an ongoing area of ecological research, but Mann & Bidwell (2001) observed acute toxicity in tadpoles exposed to agricultural surfactants.

3.3   Climate change

Climate change interacts with fire regimes to threaten biodiversity through two pathways, firstly by increasing pressures, and secondly by reducing resilience of species and ecological communities. These pathways may act individually or in combination, depending on species and ecosystem context.

In the pressures pathway, climate change is driving changes in fire regimes that increase risks of species extinction and ecosystem collapse, particularly through mechanisms of direct threat related to climate-driven changes in fire frequency (HFF), severity (HSF, LSF) and season (OSF), and the spatial patterns of fire regimes in landscapes (FSP), including the extent of individual fire events (Kelly et al., 2020). Climate change is altering fire regimes through changes in landscape dryness, heatwave incidence, wind speeds, dry lightning storms and effects on vegetation growth through changed rainfall, temperatures, atmospheric CO_2, and cyclone

severity. Anthropogenic climate change is causing more extreme weather conditions for bushfires in Australia, based on observations extending over past decades (Dowdy, 2018; Harris & Lucas, 2019; Van Oldenborgh et al., 2021), based on modelling projections of the future climate (Clarke & Evans, 2019; Dowdy et al., 2019). For example, the mean annual Forest Fire Danger Index (a widely used indicator of fire weather) is projected to increase by up to 26% by 2080 in south-eastern Australia (Clarke & Evans, 2019). In addition, rainfall and heatwave patterns affect the incidence of large fires through their effects on moisture levels in vegetation, plant litter and soils, and their effects on fire spread (Clarke et al., 2019; Nolan et al., 2016).

The changes in fire weather and increased incidence of pre-ignition droughts are facilitating fire spread into areas that over past millennia had rarely been dry enough to burn. Exampes include, rainforest and alpine vegetation within the Tasmanian Wilderness World Heritage Area (Bowman et al., 2021b) and the Gondwanan rainforest remnants, including **Critically Endangered** Lowland Rainforest of Subtropical Australia (Kooyman et al., 2020). Climate change is thus exposing these and other rainforests to direct fire-related threats, including high frequency fire (HFF) and high severity fire (HSF). Even though rainforests generally burn at substantially lower frequencies and at lower severity than nearby eucalypt forests and heathlands, they are more susceptible to degradation from low severity fires. For example, scorch heights of up to 3 m resulted in high rates of top kill for a wide range of rainforest tree species in northeast NSW and southeast Queensland during the 2019-20 fire season (Kooyman, unpubl. data). There were also high observed and estimated population declines in fauna species that depend wholly or partly on plant canopy components of unburnt rainforest such as *Sericornis citreogularis citreogularis* (Yellow-throated Scrubwrens), *Phoniscus papuensis* (Golden-tipped Bats), *Euastacus dalagarbe* (Mud Gully Crayfish) and the **Endangered** *Philoria richmondensis* (Richmond Range Sphagnum Frog) (Legge et al., 2022). Similarly, *Sphagnum cristatum* (Sphagnum Moss) and *Empodisma minus* (Spreading Rope Rush) vegetation in **Endangered** Alpine Sphagnum Bogs and Associated Fens is becoming flammable more often during droughts of increasing intensity and duration, exposing the **Critically Endangered** Corroboree frogs (Table 2) to increasing fire frequencies (HFF) and peat fires (PF) associated with the drying climate (Hunter et al., 2009; NSW Office of Environment and Heritage, 2012).

Changes to the Southern Annular Mode and weakening and occasional breakdown of the Antarctic Polar Vortex are altering wind and atmospheric moisture in the south-eastern spring and early summer (Lim et al., 2019). With reduced rainfall, this is increasing lightning incidence, and shifting the season of 'dry' lightning storms, causing extensive wildfire activity in southern Australia (Bates et al., 2018; Holz et al., 2020; Mariani et al., 2018; Styger et al., 2018). These factors have contributed to repeated fire impacts in 2003 and 2020 (HFF) on the Windswept Feldmark in the Australian Alps bioregion, currently listed in NSW as **Critically Endangered** (NSW Threatened Species Scientific Committee, 2018).

In northwestern Australia, monsoonal rainfall has increased since the early 19th century (Dey et al., 2019; Gallego et al., 2017), and there is a strong relationship between monsoonal summer rainfall, grass biomass and fire activity (Harris et al., 2008). A wide range of savanna biota show negative responses to high frequency fire (HFF) and to large fire extents (FSP), which are likely to increase due to the effect of increasing rainfall on grass fuels (Andersen, 2021).

In the second pathway, climate change is reducing the resilience of species and ecosystems so that they are less equipped to persist through, or recover from fire events. This second process

operates through indirect mechanisms of fire-related threat, such as fire-drought interactions (FDR), fire-hydrological interactions (FHY) and range of biotic interactions (FHF, FHF, FPI, FHI, FCI, FDI).

Warming and drying is slowing growth rates and reducing fecundity of some woody plants (Choat et al., 2018), reducing their ability to recover structure, develop fire resistant organs (such as thick bark or lignotubers) and build reproductive capacity before fire recurrence (FDR). These trends also extend the flammable regrowth period in tall forests following disturbance, promoting fire feedback mechanisms (Prior & Bowman, 2014; Zylstra, 2018a).

Greater frequency and severity of heat waves is exceeding temperature thresholds of physically dormant plant seed banks, depleting the numbers of seeds available for recruitment after fire events and limiting the ability of populations to recover from high levels of fire-related mortality (Ooi et al., 2009; 2012; 2014). Similar risks of recruitment failure stem from post-fire droughts (FDR) as they become more frequent and severe under a changing climate (Burgman & Lamont, 1992).

For birds, mammals and other fauna, extended droughts reduce population sizes and body condition of individuals, sometimes with substantial lags in population recovery after rains arrive (Bennett et al., 2014; Dickman et al., 2014; Recher et al., 2009). As droughts become more severe and longer under a changing climate, there is greater likelihood that fires occur when individuals are in poor condition or physiologically stressed and when population numbers are already low, increasing the risks of extinction.

The two threat pathways act in concert through processes such as 'interval squeeze', whereby climate drives increased pressure via higher fire frequency, while also reducing resilience via slower rates of maturation and lower fecundity (Enright et al., 2015). Interval squeeze potentially threatens a range of plant taxa such as *Banksia hookeriana* (Hooker's Banksia), which is at risk of local extinctions through disruption of life cycle processes (HFF and FDR, Figure 1).

Table 2 Examples of species and ecological communities threatened by the fire-related threat mechanisms identified in Figure 1 and main text

Taxon	Biota threatened	Current status under EPBC Act	Threat syndrome, sensitivity to mechanism, type of evidence	KTP criteria	References
Ecological Communities	Alpine ash forests in the Australian Alps bioregion	not currently listed	HFF High fire frequency, high sensitivity, direct evidence. HSF High severity fire, high sensitivity, direct evidence. FDR Fire-Drought interactions, medium sensitivity, suspected evidence.	a	Bowman et al., (2014); Fairman et al. (2016)
Ecological Communities	Alpine Sphagnum bogs and associated fens	EN	FHF disruption of Fire-Habitat Facilitation, high sensitivity, direct evidence.	c	Good et al. (2010)
Ecological Communities	Arnhem Plateau Sandstone Shrubland Complex	EN	HFF High fire frequency, high sensitivity, direct evidence.	c	Russell-Smith et al. (1998)

Taxon	Biota threatened	Current status under EPBC Act	Threat syndrome, sensitivity to mechanism, type of evidence	KTP criteria	References
Ecological Communities	Banksia Woodland of the Swan Coastal Plain	EN	FSP Fire Spatial Pattern, high sensitivity, suspected evidence. FCI Fire-Competition Interactions, high sensitivity, direct evidence. FDI Fire-Disease Interactions, high sensitivity, suspected evidence	c	Milberg and Lamont (1995); Moore et al. (2014)
Ecological Communities	Coastal Upland Swamps in the Sydney Basin Bioregion	EN	LFF Low fire frequency, high sensitivity, direct evidence. FCI Fire-Competition Interactions, high sensitivity, direct evidence.	c	Keith et al. (2007)
Ecological Communities	Cumberland Plain Shale Woodlands and Shale-Gravel Transition Forest	CR	LFF Low fire frequency, medium sensitivity, inferred evidence. OSF Out-of-season fire, high sensitivity, suspected evidence. FSP Fire Spatial Pattern, high sensitivity, suspected evidence. FCI Fire-Competition Interactions, high sensitivity, inferred evidence.	c	Hill and French (2003)
Ecological Communities	Eastern Stirling Range Montane Heath and Thicket	EN	HFF High fire frequency, high sensitivity, direct evidence. FHI Fire-Herbivore Interactions, medium sensitivity, inferred evidence. FDI Fire-Disease Interactions, high sensitivity, direct evidence.	b,c	Barrett and Yates (2015)
Ecological Communities	Lowland Rainforest of Subtropical Australia	CR	HFF High fire frequency, high sensitivity, inferred evidence. HSF High severity fire, high sensitivity, direct evidence. FDR Fire-Drought interactions, medium sensitivity, suspected evidence. FFL Fire-Fragmentation (Logging) interactions, high sensitivity, inferred evidence.	c	Kooyman et al. (2020); Tozer and Keith (2021)
Ecological Communities	Monsoon vine thickets on the coastal sand dunes of Dampier Peninsula	EN	HFF High fire frequency, high sensitivity, direct evidence.	c	Black et al. (2010); Lemon (2020)

Fire regimes that cause declines in biodiversity as a key threatening process

Taxon	Biota threatened	Current status under EPBC Act	Threat syndrome, sensitivity to mechanism, type of evidence	KTP criteria	References
Ecological Communities	Natural Temperate Grassland of the Victorian Volcanic Plain	CR	LFF Low fire frequency, high sensitivity, direct evidence. FSP Fire Spatial Pattern, high sensitivity, suspected evidence. FHI Fire-Herbivore Interactions, high sensitivity, inferred evidence. FCI Fire-Competition Interactions, high sensitivity, direct evidence.	c	Lunt and Morgan (2002); Morgan et al. (2017)
Ecological Communities	Swamp Sclerophyll Forest on Coastal Floodplains	not currently listed (EN, NSW)	FDI Fire-Disease Interactions, high sensitivity, inferred evidence.	c	Pegg et al. (2020)
Ecological Communities	Temperate Highland Peat Swamps on Sandstone; & Coastal Upland Swamps in the Sydney Basin Bioregion	EN EN	FHF disruption of Fire-Habitat Facilitation, high sensitivity, direct evidence. FHY Fire-Hydrological change interactions, high sensitivity, direct evidence.	c	Keith et al. (2020b)
Amphibian species	*Pseudophryne corroboree* and *Pseudophryne pengilleyi* (Corroboree frogs)	CR CR	FHF disruption of Fire-Habitat Facilitation, high sensitivity, direct evidence.	b,c	Hunter et al. (2009); NSW (2012)
Amphibian species	*Philoria richmondenis* (Richmond Range Sphagnum Frog)	EN	HFF High fire frequency, high sensitivity, direct evidence. FHF disruption of Fire-Habitat Facilitation, high sensitivity.	c	Heard et al. (2021)
Amphibian species	*Philoria sphagnicola* (Sphagnum Frog)	Not currently listed (VU in NSW)	HFF High fire frequency, high sensitivity, direct evidence. FHF disruption of Fire-Habitat Facilitation, high sensitivity.	a	–
Fish species	*Macquaria australsica* (Macquarie Perch)	EN	FDR Fire-Drought interactions, high sensitivity, inferred evidence. FHY Fire-Hydrological change interactions, high sensitivity, direct evidence.	c	Baumgartner et al. (2020)
Fish species	*Maccullochella macquariensis* (Trout Cod)	EN	FDR Fire-Drought interactions, high sensitivity, inferred evidence. FHY Fire-Hydrological change interactions, high sensitivity, direct evidence.	c	Silva et al. (2020)
Bird species	*Malurus melanocephalus* (Red-backed Fairy-wren)	not currently listed	OSF Out-of-season fire, medium sensitivity, direct evidence. FPI Fire-Predator Interactions, medium sensitivity, direct evidence.	a	Murphy et al. (2010)

Taxon	Biota threatened	Current status under EPBC Act	Threat syndrome, sensitivity to mechanism, type of evidence	KTP criteria	References
Bird species	*Manorina melanotis* (Black-eared Miner)	EN	HFF High fire frequency, high sensitivity, suspected evidence. FSP Fire Spatial Pattern, medium sensitivity, inferred evidence.	c	Taylor et al. (2013)
Bird species	*Pezoporus flaviventris* (Western Ground Parrot, Kyloring)	CR	HSF High severity fire, high sensitivity, suspected evidence. FPI Fire-Predator Interactions, high sensitivity, direct evidence.	c	Burbidge et al. (2016); Comer et al. (2020)
Bird species	*Phylidonyris niger* (White-cheeked Honeyeater)	not currently listed	HFF High fire frequency, high sensitivity, direct evidence. FHF disruption of Fire-Habitat Facilitation, high sensitivity, direct evidence.	a	Brooker and Rowley (1991)
Bird species	*Psephotus chrysopterygius* (Alwal, Golden-shouldered Parrot)	EN	LFF Low fire frequency, medium sensitivity, direct evidence. FHF disruption of Fire-Habitat Facilitation, medium sensitivity, direct evidence. FHI Fire-Herbivore Interactions, medium sensitivity, direct evidence.	b,c	Bateman and Johnson (2011); Garnett and Crowley (2002); Reside et al. (2016)
Bird species	*Polytelis alexandrae* (Princess Parrot)	VU	HSF High severity fire, high sensitivity, suspected evidence. FHF disruption of Fire-Habitat Faciliation, high sensitivity, direct evidence.	b,c	Pavey et al. (2014)
Bird species	*Stipiturus mallee* (Mallee Emu-wren), *Pachycephala rufogularis* (Red-lored Whistler), *Psophodes nigrogularis leucogaster* (Western Whipbird)	EN VU VU	HFF High fire frequency, medium sensitivity, suspected evidence. OSF Out-of-season fire, medium sensitivity, inferred evidence. FPI Fire-Predator Interactions, medium sensitivity, suspected evidence.	c	Department of Environment Land Water and Planning (2016)
Reptile species	*Liopholis kintorei* (Great Desert Skink)	VU	FSP Fire Spatial Pattern, medium sensitivity, direct evidence. FPI Fire-Predator Interactions, high sensitivity, direct evidence.	c	McAlpin (2001); Moore et al. (2018); Paltridge et al. (2020)
Mammal species	*Antechinus minimus maritimus* (Swamp Antechinus (mainland))	VU	HFF High fire frequency, High sensitivity, high sensitivity, direct evidence. FDR Fire-Drought, high sensitivity, direct evidence. FFC Fire-Fragmentation (Clearing) interactions, high sensitivity, suspected evidence.	a	Wilson et al. (2001); Wilson et al. (2017)

Taxon	Biota threatened	Current status under EPBC Act	Threat syndrome, sensitivity to mechanism, type of evidence	KTP criteria	References
Mammal species	*Bettongia penicillata ogilbyi* (Woylie)	EN	FPI Fire-Predator Interactions, High sensitivity.	c	Christensen (1980)
Mammal species	*Bettongia tropica* (Northern Bettong)	EN	FPI Fire-Predator Interactions, high sensitivity, inferred evidence. FHI Fire-Herbivore Interactions, high sensitivity, suspected evidence. FCI Fire-Competition Interactions, high sensitivity, inferred evidence.	c	Whitehead et al. (2018)
Mammal species	*Gymnobelideus leadbeateri* (Leadbeater's Possum)	CR	HFF High frequency fire, high sensitivity, direct evidence. HSF High severity fire, high sensitivity, direct evidence. FHF disruption of Fire-Habitat Facilitation, high sensitivity, direct evidence. FSP Fire Spatial Pattern, high sensitivity, direct evidence. FFL Fire-Fragmentation (Logging) interactions, high sensitivity, direct evidence.	c	Lindenmayer et al. (2021); Nitschke et al. (2020); Threatened Species Scientific Committee (2019)
Mammal species	*Isoodon obesulus* (Southern Brown Bandicoot (south-eastern))	EN	FPI Fire-Predator Interactions, High sensitivity.	c	Robley et al. (2017)
Mammal species	*Lagorchestes hirsutus Central Australian subspecies* (Mala, Rufous Hare-Wallaby (Central Australia))	EN	FSP Fire Spatial Pattern, medium sensitivity, inferred evidence. FPI Fire-Predator Interactions, high sensitivity, suspected evidence.	c	Lundie-Jenkins (1993); Short and Turner (1994)
Mammal species	*Ningaui yvonneae* (Mallee Ningaui)	not currently listed (VU, NSW)	FHF disruption of Fire-Habitat Facilitation, medium sensitivity, direct evidence. FSP Fire Spatial Pattern, medium sensitivity, direct evidence. FPI Fire-Predator Interactions, high sensitivity, inferred evidence.	a	Kelly et al. (2011, 2012)

Taxon	Biota threatened	Current status under EPBC Act	Threat syndrome, sensitivity to mechanism, type of evidence	KTP criteria	References
Mammal species	*Petauroides volans* (Greater Glider)	VU	HFF High fire frequency, high sensitivity, direct evidence. HSF High severity fire, high sensitivity, direct evidence. FHF disruption of Fire-Habitat Facilitation, high sensitivity, direct evidence. FSP Fire Spatial Pattern, high sensitivity, inferred evidence. FPI Fire-Predator Interactions, high sensitivity, direct evidence. FFL Fire-Fragmentation (Logging) interactions, high sensitivity, direct evidence.	b,c	Andrew et al. (2014); Berry et al. (2015); Chia et al. (2015a); Kavanagh (1992); Lindenmayer et al. (2018; 2013); Lunney (1987); McLean et al. (2018); van der Ree and Loyn (2002)
Mammal species	*Phascolarctos cinereus* (Koala (combined populations of Qld, NSW and ACT)	EN	HSF High severity fire, high sensitivity, direct evidence. FPI Fire-Predator Interactions, high sensitivity, direct evidence. FDR Fire-Drought interactions, high sensitivity, direct evidence. FFC Fire-Fragmentation (Clearing) interactions, High sensitivity.	b,c	Lane et al. (2020); Law et al. (2022); Lunney et al. (2007; 2014); Matthews et al. (2016); Phillips et al. (2021);
Mammal species	*Potorous gilberti* (Gilbert's Potoroo, Ngilkat)	CR	FHF disruption of Fire-Habitat Facilitation, high sensitivity, direct evidence. FSP Fire Spatial Pattern, high sensitivity, direct evidence. FPI Fire-Predator Interactions, high sensitivity, inferred evidence.	–	DAWE (2021a); DPW (2016)
Mammal species	*Zyzomys pedunculatus* (Central Rock-rat, Antina)	CR	HFF High fire frequency, high sensitivity, direct evidence. FPI Fire-Predator Interactions, high sensitivity, suspected evidence. FDR Fire-Drought interactions, high sensitivity, suspected evidence.	b,c	McDonald et al. (2016); Nano et al. (2019)
Invertebrate species	*Castiarina insculpta* (Miena Jewel Beetle)	not currently listed (EN, TAS)	HFF High fire frequency, high sensitivity, inferred evidence. HSF High severity fire, high sensitivity, inferred evidence. FHF disruption of Fire-Habitat Facilitation, high sensitivity, inferred evidence. FSP Fire Spatial Pattern, high sensitivity, inferred evidence. FSD Fire-Suppression Disturbance, High sensitivity.	–	–

Taxon	Biota threatened	Current status under EPBC Act	Threat syndrome, sensitivity to mechanism, type of evidence	KTP criteria	References
Invertebrate species	*Zephyrarchaea austini* (Kangaroo Island Assassin Spider)	currently under assessment for listing	HFF High fire frequency, high sensitivity, direct evidence. FHF disruption of Fire-Habitat Facilitation, high sensitivity, inferred evidence.	a	Marsh and Glatz (In review)
Invertebrate species	*Pseudococcus markharveyi* (Banksia montana mealybug)	CR	HFF High fire frequency, high sensitivity, direct evidence. FHF disruption of Fire-Habitat Facilitation, high sensitivity, inferred evidence.	b	Moir (2021)
Plant species	*Athrotaxis cupressoides* (Pencil Pine)	not currently listed	HFF High frequency fire, high sensitivity, direct evidence. HSF High severity fire, high sensitivity, direct evidence. FDI Fire-Disease Interactions, high sensitivity, inferred evidence. FSR Fire-Suppression Retardant chemicals, Medium to high sensitivity, inferred evidence.	–	Bell et al. (2005)
Plant species	*Banksia cuneata* (Matchstick Banksia, Quairading Banksia)	EN	HFF High frequency fire, high sensitivity, direct evidence. LFF Low fire frequency, high sensitivity, direct evidence. FDR Fire-Drought interactions, high sensitivity, direct evidence. FFC Fire-Fragmentation (Clearing) interactions, high sensitivity, direct evidence.	b,c	Burgman and Lamont (1992)
Plant species	*Banksia montana* (Stirling Range Dryandra)	EN	HFF High fire frequency, high sensitivity, direct evidence. OSF Out-of-season fire, high sensitivity, inferred evidence. FDI Fire-Disease Interactions, high sensitivity, direct evidence.	b,c	Barrett et al. (2008); Gilfillian et al. (2005)
Plant species	*Epacris barbata* (Bearded Heath, Freycinet Heath)	EN	HFF High fire frequency, high sensitivity, direct evidence. FDI Fire-Disease Interactions, high sensitivity, direct evidence.	c	Keith (2004)
Plant species	*Grevillea caleyi* (Caley's Grevillea)	CR	HFF High fire frequency, high sensitivity, direct evidence. LSF Low severity fire, Medium to high sensitivity, inferred evidence. FSP Fire Spatial Pattern, high sensitivity, direct evidence. FFC Fire-Fragmentation (Clearing) interactions, high sensitivity, direct evidence.	b,c	Regan et al. (2003)

Taxon	Biota threatened	Current status under EPBC Act	Threat syndrome, sensitivity to mechanism, type of evidence	KTP criteria	References
Plant species	*Pomaderris walshii* (Carrington Falls Pomaderris)	CR	HFF High fire frequency, medium sensitivity. LSF Low severity fire, medium to high sensitivity.	b,c	le Breton et al. (2020)
Plant species	*Leucopogon exolasius* (Woronora Beard-heath)	VU	HFF High fire frequency, medium sensitivity, inferred evidence. OSF Out-of-season fire, medium sensitivity, direct evidence. FDI Fire-Disease Interactions, medium sensitivity, suspected evidence. FDR Fire-Drought interactions, medium sensitivity, suspected evidence.	b,c	Ooi et al. (2007)
Plant species	*Rutidosis leptorhynchoides* (Button Wrinklewort)	EN	LFF Low fire frequency, high sensitivity, direct evidence. OSF Out-of-season fire, high sensitivity, direct evidence. FCI Fire-Competition Interactions, high sensitivity, direct evidence. FFC Fire-Fragmentation (Clearing) interactions, high sensitivity, direct evidence.	c	Morgan (1995)
Plant species	*Pterostylis basaltica* (Basalt Greenhood)	EN	OSF Out-of-season fire, medium sensitivity, suspected evidence. FHI Fire-Herbivore Interactions, high sensitivity, inferred evidence.	c	Ingeme and Backhouse (1999)
Plant species	*Verticordia fimbrilepis* subsp. *Fimbrilepis* (Shy Featherflower)	EN	LFF Low fire frequency, high sensitivity, direct evidence. FFC Fire-Fragmentation (Clearing) interactions, high sensitivity, direct evidence.	b,c	Yates and Ladd (2010)

Type of evidence is categorised as direct (e.g., from published experiments or field studies), inferred (e.g., from similar taxa or communities), suspected (e.g., from life history traits). FSP Fire Spatial Pattern includes low or high patchiness; FCI Fire-Competition Interactions includes post-fire weed invasion. KTP criteria refers to the criteria (a, b, c) described in section 4. How judged by the Committee in relation to EPBC Act criteria in Section 188(4) of the EPBC Act. Current status in the EPBC Act column is based on the Species Profile and Threats Database (SPRAT) at 26 October 2021.

4   How judged by the Committee in relation to EPBC Act criteria

Section 188(4) of the EPBC Act states:

> *A threatening process is eligible to be treated as a key threatening process if:*
>
> *(a)   it could cause a native species or an ecological community to become eligible for listing in any category, other than conservation dependent; or*
>
> *(b)   it could cause a listed threatened species or a listed threatened ecological community to become eligible to be listed in another category representing a higher degree of endangerment; or*
>
> *(c)   it adversely affects 2 or more listed threatened species (other than conservation dependent species) or 2 or more listed threatened ecological communities.*

'Fire regimes that cause declines in biodiversity' meet all three criteria. Evidence supporting the eligibility for listing based on each criterion is given in examples with supporting references in Table 2, with specific case studies summarised below.

Criterion (a): Could the process cause a native species or an ecological community to become eligible for listing as Extinct, Extinct in the Wild, Critically Endangered or Vulnerable?

There are many species and ecological communities not currently listed as 'threatened' under the EPBC Act that are likely to be to become eligible for listing as a consequence of fire-related processes described in this Advice. For example, a large number of species and ecological communities became be eligible for listing as threatened, or are being considered for listing as threatened, after the 2019-20 megafires in southern Australia, including:

Vertebrates: Over 40 vertebrate species were identified as potentially eligible for listing as threatened by the Wildlife and Habitat Bushfire Recovery Expert Panel, and a related National Environmental Science Program research project (Legge et al., 2020; 2021a; 2021b; 2022). In many cases, species were declining due to several threats, often interacting with fire via one of the mechanisms depicted in Fig. 1, such as High Frequency Fire (HFF), High Severity Fire (HSF), Fire-drought interactions (FDR), Fire Spatial Pattern (FSP), Fire-predator interactions (FPI) and Fire-Herbivore Interactions (FHI).

Invertebrates: At least 80 species were identified as potentially eligible for listing as threatened (Legge et al., 2020; 2022; Marsh et al., 2021; Woinarki 2020), as a result of mechanisms such as High Frequency Fire (HFF), High Severity Fire (HSF) Fire-drought interactions (FDI).

Plants: a national analysis after the fires identified 486 plant species from 143 genera that were a high priority for survey and management after the fires. Approximately 200 of these plant species are potentially eligible for listing as threatened as a result of high fire frequency (HFF), high fire severity (HFS) and several interactions such as those between fire and herbivory (FHI), and fire and drought interactions (FDR) both before and after the fires (Gallagher, 2020; Gallagher et al., 2021).

Ecological communities: At least 52 fire-affected ecological communities were identified as candidates for listing as threatened, including 30 that are currently listed under state or territory legislation. There are 53 ecological communities not currently listed as threatened that are in need of high priority management for recovery after the 2019-20 bushfires. (Keith et al., 2021). The most common mechanisms of threat to ecological communities include High Frequency Fire (HFF); Fire-drought interactions (FDI); and Fire-predator interactions (FPI) (Keith et al., 2022).

Some fungi species are also threatened by inappropriate fire regimes, but in general the evidence base is too poor to support listing assessments (Dell et al., 2020; McMullan-Fisher et al., 2011).

Two examples of unlisted entities that are likely to be to become eligible for listing as a consequence of fire-related processes described in this Advice are elaborated below. These examples were not affected by the 2019-20 megafires.

- Mallee Ningaui – *Ningaui yvonneae*, and

- Windswept feldmark in the Australian Alps bioregion.

Mallee Ningaui – *Ningaui yvonneae*

The Southern or Mallee Ningaui is a small (4-10g) carnivorous dasyurid of mallee ecosystems with an understorey of hummock grass (*Triodia*) or shrubs. It is found in semi-arid southern Australia across WA, SA, Vic and NSW (Strahan, 1995). Currently listed in NSW as **Vulnerable** the Mallee Ningaui is threatened by frequent fire, feral predators, land clearing and grazing (NSW Office of Environment and Heritage, 2017).

Spinifex (*Triodia* spp.) is a primary fuel for bushfire spread in these ecosystems (Haslem et al., 2011) and the primary habitat for Mallee Ningaui (Bos et al., 2002). Fire has strong impacts on Mallee Ningaui populations (Kelly et al., 2011; 2012). Work to date has focused on the indirect and interactive effects of fire regimes which have the potential to cause the Mallee Ningaui to become threatened via alteration of the spatial and temporal availability of spinifex habitat (Kelly et al., 2011; 2012). Spinifex is important to Mallee Ningaui both for cover and as a hunting environment (Bos & Carthew, 2007), and Mallee Ningaui numbers correlate with spinifex cover. Numbers are low within five years after fire, but increase thereafter as spinifex cover is restored (Kelly et al., 2011). Greater cover of spinifex increases the likelihood of fire (Burrows et al., 2018), but this begins to decline after ~30-40 years (Haslem et al., 2011), once again reducing fire risk. Spinifex hummocks of this age and older are, however, of greatest habitat value to Mallee Ningaui as they collapse at the centre to form rings. Mallee Ningaui numbers – in particular the number of breeding females – remain high beyond this period (Kelly et al., 2020).

Habitat can therefore be divided into three age categories (Kelly et al., 2011). Young (up to five years since fire) habitat supports very few animals but has a low risk of fire. Regrowth habitat (5 – 40 years old) supports growing populations, but also has the maximum risk of fire; and mature habitat (>40 years old) has both the highest habitat value, and the lowest risk of fire. These changes may interact with predation by feral cats and foxes, as work on other species indicates that reduced cover following fire increases predation of small mammals (Hradsky, 2020; Leahy et al., 2016). The strong preference for more mature vegetation makes it likely that Mallee Ningaui utilise a 'habitat supplementation' movement pattern (Nimmo et al., 2018), so that large areas of young, recently burnt spinifex fragment their habitat (Kelly et al., 2020).

This requirement for relatively closely located or connected areas of long-unburnt habitat renders Mallee Ningaui vulnerable to potential changes in fire regimes, specifically, an increase in the frequency of large fires. Since 2009, a focus on increasing the area of prescribed burning has resulted in increases in the amount of Victorian mallee environments burnt (Inspector-General for Emergency Management, 2015). Pressure exists to burn remote areas as these are cheaper to burn (Florec et al., 2020), and the cost per hectare of prescribed burning is currently a Key Performance Indicator for some states (DBCA, 2019).

Windswept feldmark in the Australian Alps bioregion

Windswept feldmark is a highly restricted (28.5 ha) ecological community composed of dwarf plants dominated by a prostrate growth form of the shrub *Epacris gunnii*, and limited to the most exposed sites between 2010 – 2150 m above sea level on the Main Range of NSW. Feldmark is currently listed in NSW as **Critically Endangered** due to threats related to climate change, trampling by hikers and feral animals such as horses, and fire (NSW Threatened Species Scientific Committee, 2018). It is not yet listed as a threatened ecological community under the EPBC Act.

Windswept feldmark is particularly vulnerable to fire for two reasons. Firstly, all patches of the community are located in areas that are exposed to the strongest westerly winds at the top of a very steep, west-facing slope. This locates the patches directly in the path of severe fires. Secondly, the community is dependent upon the shrub *Epacris gunnii* as a keystone species and ecosystem engineer (Ballantyne & Pickering, 2015). The slow growth of this species (~1 cm y^{-1}, Barrow et al., 1968) means that fire damage will have very long-lasting effects.

The risks posed by fires have increased due to climate change, accentuated by positive flammability feedbacks in the vegetation on the slopes below the patches. Historic fire impacts on Windswept feldmark were entirely anthropogenic in origin, burning up the steep western slopes of the Main Range in 1939 and 1952 (Zylstra, 2006). Although that threat was managed by tighter Government controls on fire, shifts in the Southern Annular Mode and the El Niño Southern Oscillation have increased the incidence of dry lightning storms causing summer ignitions coupled with drought conditions that remove natural moisture-based controls on fire spread (Boer et al., 2017; Mariani et al., 2016, 2018; Mariani & Fletcher, 2016; Nolan et al., 2016). In addition to these factors, the Antarctic Polar Vortex has been weakening since 2000, resulting in its partial collapse in 2002-03 and significant weakening in 2019-20 (Byrne & Shepherd, 2018; Lim et al., 2018; 2019; 2020). Weakening of the vortex causes increased westerly wind flow over south-eastern Australia, and both collapse events coincided with dry lightning events igniting fires that, fanned by those winds, impacted directly on the Windswept feldmark communities. Amplifying these effects, positive feedbacks in forests and treeless

communities below the feldmarks have increased the likelihood of fire spreading through those areas (Camac et al., 2017; Zylstra, 2018a).

Conclusion for Criterion (a)

The Committee considers that the threatening process is eligible under this criterion as the process could cause multiple species and ecological communities, including but not limited to the Mallee Ningaui and the Windswept feldmark in the Australian Alps bioregion, and the additional species and ecological communities identified as meeting criterion (a) in Table 2, to become eligible for listing as threatened under the EPBC Act.

Criterion (b): Could the threatening process cause a listed species or a listed ecological community to become eligible to be listed in another category representing a higher degree of endangerment?

Many species and a number of ecological communities currently listed as threatened could become eligible for listing in a higher category of threat as a result of 'Fire regimes that cause declines in biodiversity'.

For example, over 15 threatened vertebrate species, at least three invertebrate species, over 40 plant species and two ecological communities were identified as potentially eligible for uplisting by the Wildlife and Habitat Bushfire Recovery Expert Panel, and related National Environmental Science Program research projects (Gallagher, 2020; Gallagher et al., 2021; Keith et al., 2020b; 2022; Legge et al., 2022; 2021a; 2021b; Marsh et al 2021) and many plant species were subsequently formally prioritised for reassessment by the committee in the 2020 and 2021 Finalised Priority Assessment Lists (DAWE, 2020; 2021a). In many cases, species were declining due to several threats, often interacting with fire via one of the mechanisms depicted in Fig. 1.

Two examples of listed entities that are likely to be eligible for uplisting as a consequence of fire-related processes described in this Advice are elaborated below. These examples were both affected by the 2019-20 megafires.

- Eastern Stirling Range Montane Heath and Thicket, and

- Greater Glider – *Petauroides Volans.*

Eastern Stirling Range Montane Heath and Thicket

Eastern Stirling Range Montane Heath and Thicket is an **Endangered** treeless community restricted to <500 ha of the Stirling Ranges in south-western Australia, recognised as **Critically Endangered** by the WA Threatened Ecological Communities Advisory Committee. Root rot disease caused by *Phytophthora cinnamomi* is currently recognised as the primary threat to the community, although frequent fire, recreation, and grazing by rabbits are also identified (Barrett, 2000; Threatened Species Scientific Committee, 2017).

The documented fire history of the community includes a fire in 1972 and a second fire in 1991 (Barrett, 2000). An overlay of the NVIS mapped area of the community (DAWE, 2020) with the mapped fire history (DBCA, 2018) indicates that, since these earlier events, the community has also been affected by fires in 1996, 1997, 2000, 2010, 2018 and 2019. Data presented by Keith et al. (2014) and Barrett & Yates (2015) indicate substantial declines in key shrub species of the

community occurred after the 2000 fire following a relative short fire interval of 9 years since the preceding fire (section 3.1.1 High frequency fires, HFF). Keith et al. (2014) suggest that disease and fire (section 3.2.1.5, FDI) acted synergistically to accelerate the collapse of obligate-seeding populations of Proteaceae shrubs, while other shrub taxa (e.g., Ericaceae) also exhibit rapid declines. They concluded that long-term monitoring of the community demonstrated dramatic changes resulting in a less diverse thicket community, albeit still with significant conservation values. Since then, more than 93% of the community was burnt in separate fires in 2018 and late 2019, including an area burnt in both fire events (Keith et al., 2022). The ecological outcomes are currently being studied, but include the likely extinction of a local endemic hemipteran insect dependent on a single plant host species within the community (Moir, 2021). Given the direct effects of fire already documented (Barrett, 2000; Threatened Species Scientific Committee, 2017) along with the interactive effects of fire and *P. cinnamomi*, the ecological community is at greater risk due to increases in fire frequency.

Greater Glider – *Petauroides volans*

The Greater Glider *Petauroides volans* is the largest gliding possum in Australia. It occurs in eastern Australian forests from around Proserpine in Queensland to central and southern Victoria, where it feeds on the foliage, flowers and buds of a narrow range of *Eucalyptus* species and dens in tree hollows. It was listed as Vulnerable in 2016 (the listing included populations in northeast Qld, now considered to be a separate species, the northern greater glider *Petauroides minor* and populations to the south which continue to be recognised as *Petauroides volans*. Both taxa are being considered for uplisting as part of the Committee's reassessment of the status of *Petaurodes volans* which was prioritised in response to the 2019-20 bushfires (DAWE, 2020)).

The Greater Glider is declining throughout its range (Threatened Species Scientific Committee, 2016c). Fire is contributing to these declines in several ways. High-severity fires (section 2.1.3, HSF) cause substantial direct mortality due to lethal heating and suffocation (Berry et al., 2015) and indirect mortality due to loss of critical resources, especially den sites and food. Abundance of Greater Gliders typically falls immediately after major fires (Lindenmayer et al., 2013). Because Greater Gliders have low rates of reproduction, populations are slow to recover from single fire events. Therefore, high-frequency fire (section 3.1.1, HFF) can have large cumulative effects that cause long-term decline.

Fire interacts with other threats to amplify impacts on Greater Gliders. The species is threatened by loss of old hollow-bearing trees due to changes in forest structure caused by timber harvesting (Lindenmayer et al., 2017). Fires disrupt habitat facilitation (section 3.2.1.1, FHF) by accelerating these declines by weakening old trees and causing them to collapse. Following the 2009 fires in Victoria, 79% of large hollow-bearing trees in mountain ash forest died (Lindenmayer et al., 2013). Removal of mature hollow-bearing trees in silviculture management may increase flammability by stimulating growth of dense understorey vegetation and increasing exposure of fuels to drying, resulting in further impacts of fire on Greater Glider habitat (Fire-fragmentation interactions and other effects associated with logging/timber harvesting, FFL). The Greater Glider is sensitive to the effects of forest fragmentation because individuals have limited ability to cross cleared land between forest patches (Pope et al., 2005). Therefore, patches in fragmented landscapes may not be recolonised once extirpated by disturbances such as fire (Fire-fragmentation effects due to clearing, FFC). Because of the species' strict eucalypt diet and associated unique physiology, individual Greater Gliders are susceptible to stress from high temperature and low leaf-moisture content in drought (Moore et

al., 2004). High temperature and drought also increase the likelihood of high-severity fire, leading to greater combined mortality (Fire-drought interactions, FDI).

The 2019-2020 megafires affected approximately 35% of the distribution of the Greater Glider; much of this area was burnt at high severity. The full impact of these fires on the Greater Glider is not yet known but an analysis using expert elicitation suggested that the Greater Glider population would decline by 33% over three generations as a result of the megafires, additional to continuing effects of other threats (Threatened Species Recovery Hub, 2021). It is therefore likely that recent declines of the Greater Glider have been accelerated by fire, and that as a result the species will be eligible for listing in a higher category of threat in the near future.

Conclusion for Criterion (b)

The Committee considers that the threatening process is eligible under this criterion as the process could cause multiple species and ecological communities, including but not limited to the **Endangered** Eastern Stirling Range Montane Heath and Thicket ecological community, the **Vulnerable** Greater Glider, and the additional species and ecological communities identified as meeting criterion (b) in Table 2, to become eligible under this criterion to be listed in a higher threat category.

Criterion (c): Does the threatening process adversely affect two or more listed threatened species (other than conservation dependent species) or two or more listed threatened ecological communities?

More than 800 native species and 65 ecological communities listed under Australian legislation as at December 2020 (DAWE, 2021a) are identified in Conservation Advice as threatened by inappropriate fire regimes. Examples of 23 threatened species and 11 threatened ecological communities affected by multiple direct or indirect mechanisms of fire-related threat are identified in Table 2. Three examples are elaborated:

- Colquhoun Grevillea – *Grevillea celata*

- Great Desert Skink – *Liopholis kintorei*

- Koala – *Phascolarctos cinereus*.

Colquhoun Grevillea – *Grevillea celata*

Colquhoun Grevillea is a **Vulnerable** root-suckering shrub which grows to approximately 1.8 m tall. Populations of Colquhoun Grevillea occur in heathy open forest with an overstorey of *Eucalyptus* species (Department of Sustainability and Environment, 2009; Walsh & Entwisle, 1996).

The species is endemic to a small area in central eastern Gippsland, Victoria (Molyneux, 1995). The total range of all known populations is 11 km^2 (Department of Sustainability and Environment, 2009). Recent surveys estimate that approximately 1500 plants occur across nine populations in the Colquhoun State Forest. In the past, surveys have estimated the species' population to be over 2300 plants (Carter & Walsh, 2006), indicating that there has been a significant decline in the species' population size. Currently many plants occur in highly insecure situations such as roadsides, where they may be at risk from processes such as road works.

However, the main threat to the species has been identified as inappropriate fire regimes (Carter & Walsh, 2006; Department of Sustainability and Environment, 2009).

In many parts of the species' range, low-severity fuel reduction burns occur frequently at short intervals, and this has resulted in high densities of fire-promoted species including *Pteridium esculentum* (bracken), which Colquhoun Grevillea does not appear to be able to tolerate. The species tends to be confined to roadsides and natural forest clearings where the vegetation is less dense and there are higher light levels (Carter & Walsh, 2006).

Colquhoun Grevillea appears to return very slowly following a fire, and the species may be severely browsed by native herbivores during its juvenile stage. The recovery plan for this species recommends that fires should be no more frequent than every ten years, as anything less will lead to a dense cover of bracken and will not provide suitable habitat for the species (Carter & Walsh, 2006). Subsequent work has however found that on average for Victorian forests, near-surface and elevated fuels reach their maximum densities at approximately this age (McColl-Gausden et al., 2020), and so less frequent fire should reduce the risk of decline of the Colquhoun Grevillea as well as achieving lower fuel loads.

Great Desert Skink – *Liopholis kintorei*

The **Vulnerable** Great Desert Skink is a large burrowing lizard that lives in social groups, in spinifex grassland sandplains (McAlpin et al., 2011). The skink plays an important ecosystem role as an upper-level predator, and its burrows are used by a variety of other species such as *Dasycercus cristicauda* (Crest-tailed Mulgara) (McAlpin, 2001).

The great desert skink is believed to have previously been distributed across a series of widespread, but connected, populations in the Great Sandy, Little Sandy, Gibson, Great Victoria and Tanami Deserts in the eastern interior of Western Australia, south-western Northern Territory and north-western South Australia (Cogger et al., 1993; McAlpin, 2001). Currently, the species occurs in a smaller area of the Western Deserts region of Central and Western Australia (Indigo et al., 2021).

Fire regimes have changed in some parts of the spinifex deserts following the loss of traditional fire practices; specifically, fire frequency and severity have increased where the fine-scale and intensive application of fire by people has been replaced by extensive fires ignited by lightning, at least during years of average rainfall (Blackwood et al., 2021; Burrows et al., 2006; Edwards et al., 2008; Wright et al., 2021a). Although these changes in the fire regime (HSS and HSF) may affect great desert skinks directly, the main impact pathway appears to be through an interaction with predators (FPI). Extensive fire removes cover from around the burrow entrances of the great desert skinks, making them more vulnerable to predation whilst foraging, basking, and moving between burrows, especially by introduced predators (red fox, cats). Conversely, small-scale fires that either prevent wildfires from reaching burrow areas, or that are internally patchy and retain cover around burrow entrances, can help skinks avoid detection from predators (Cadenhead et al., 2016; Moore et al., 2015, 2018; Paltridge 2020).

The interaction between fire and increased predation from cats, foxes and possibly dingoes has led to the widespread decline of the species (Threatened Species Scientific Committee, 2016b).

Koala – *Phascolarctos cinereus*

In 2012 the combined population of koalas in Qld, NSW and the ACT were listed under the EPBC Act as **Vulnerable**, due to declining populations (Threatened Species Scientific Committee, 2012). However, since that listing, koalas have become increasingly affected by a range of threats including inappropriate fire regimes, and interactions between fire and other threats. The koala (in Qld, NSW, ACT) was transferred to **Endangered** in February 2022 (Threatened Species Scientific Committee, 2022). Information supporting that re-assessment is summarised below.

Koalas are adversely affected by the increasing frequency (HFF) and severity of fire (HSF), driven by the changing climate. High severity fires (HSF) cause substantial immediate population losses. As a result, koalas are rare or absent in areas that experience heavy crown scorch (Law et al., 2022; Phillips et al., 2021). As well as direct mortality from the fire itself, koalas can die in the weeks after fire because of the high metabolic costs of finding forage trees in the burnt landscape (Nagy & Martin, 1985). Koalas can survive lower severity fires, especially if refuge areas such as moist gullies are available (Davies et al., 2013; Lunney et al., 2014). However, increasing fire frequencies, even of moderately severe fires, could prevent populations from recovering between fire events, because even low-moderate severity fires can cause burns and mortality for koalas (Zylstra, 2019; Zylstra et al., 2016).

Fires also interact with several other processes that threaten koalas (summarised from Threatened Species Scientific Committee, 2022). For example, the changing climate is reducing the area of suitable habitat for koalas, as they are adversely affected by drought and high temperatures. These extreme weather conditions also increase the likelihood of severe fires, and may make koalas more susceptible to fire-related impacts by lowering their condition ahead of the fire (FDR). Landscape fragmentation from clearing and logging increase the impacts of fire (FFC, FFL), by increasing the difficulty of recolonising severely burnt and isolated fragment after fire. Fragmentation also means that surviving koalas have to travel further between fragments to find unburnt habitat. Koalas are threatened by predation from wild and domestic dogs when they travel on the ground between trees or habitat patches. Fire interacts with this predation threat (FPI) by removing ground vegetation, causing surviving koalas to be more exposed to predators. Fires can also force koalas to disperse further to find food in unburnt areas, again increasing exposure to predators and also vehicle strike in peri-urban areas (FPI, FFC). Some koala populations are threatened by disease (e.g., Chlamydia), and disease prevalence increases in populations exposed to other stressors (FDI).

The direct and interactive effects of fire have contributed to making the koala eligible for transfer from Vulnerable to the Endangered category under the EPBC Act (Threatened Species Scientific Committee, 2022).

Conclusion for Criterion (c)

The Committee considers that the threatening process is eligible under this criterion as the process is adversely affecting more than two listed threatened species, including the Colquhoun Grevillea, the Great Desert Skink and the combined koala populations of Qld, NSW and the ACT, and the additional species and ecological communities identified as meeting criterion (c) in Table 2.

4.1   Public consultation

Notice of the proposed amendment to the list of key threatening processes was made available for public comment for at least 30 business days between 19 November 2021 and 10 January 2022. All comments received have been taken into account in this assessment.

4.2   Conclusion

The threatening process meets s188 (4) (a) (b) and (c) of the EPBC Act and is therefore eligible to be listed as a key threatening process.

5   Recommendations

A. The Committee recommends that the list referred to in section 183 of the EPBC Act be amended by including in the list as a key threatening process: 'Fire regimes that cause declines in biodiversity'.

Threatened Species Scientific Committee

09/03/2022

Glossary

Term	Definition
Australia	For the purpose of this nomination Australia is defined as including all of continental Australia and its immediate off-shore islands. Distant Australian territories are excluded.
Alien species	In this document the term 'alien species' refers to species that have been introduced to Australia from other countries, as well as species native to Australia that now occur outside their pre-European range.
Available fuel	The proportion of fine living or dead plant biomass that is both sufficiently dry, aerated and within reach of flames to ignite. This varies on an hourly basis for dead material, and over days and weeks for living tissues.
Biodiversity	Biodiversity is the variety of all life forms on earth; it is the different plants, animals and micro-organisms; their genes; and the terrestrial, marine and freshwater ecosystems of which they are a part.
Contemporary	Present-day or developed since European settlement and especially over the last few decades.
Ecological community	An assemblage of native species which inhabits a particular area in nature. Ecological communities are unique and naturally occurring groups of plants and animals. Their distribution and features are influenced by factors such as soil type, position in the landscape, climate and water availability.
Fire intensity	The quantity of energy produced as heat by a fire. Measured as per metre of fire front as Intensity = H.w.R, (H = energy content of fuel, w = biomass consumed, R = rate of spread). This is rarely estimated, and modelled values are unreliable as numerous studies have found that they bear little relationship to fire behaviour.
Fire regime	See section 2.1 What is a fire regime.
Fire-sensitive vegetation (or species)	A loosely applied term that refers to a species or community that is highly responsive to particular fire characteristics (e.g., readily eliminated by severe fires) or specifically to obligate-seeder plants in which standing individuals are killed outright by fire.
Fire severity	See Table 1.
First Nations	In this document, First Nations is used when referring to the era between arrival and European settlement i.e. First Nations era.
Inappropriate fire regimes	In this document, inappropriate fire regimes are fire regimes that cause declines in biodiversity.
Prescribed burning (Fuel Reduction Burning)	The controlled application of fire under specified environmental conditions to a predetermined area and at the time, intensity and rate of spread required to attain planned resource management objectives.
Regions	This Advice uses the regions identified in the Interim Biogeographic Regionalisation of Australia (IBRA).

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