Conservation Advice for *Petaurus australis australis* (yellow-bellied glider (south-eastern))

In effect under the *Environment Protection and Biodiversity Conservation Act 1999* from 2 March 2022.

This document combines the conservation advice and listing assessment for the subspecies. It provides a foundation for conservation action and further planning.

Two yellow-bellied gliders © Copyright Rohan Bilney

Conservation status

Petaurus australis australis (yellow-bellied glider (south-eastern)) is listed in the Vulnerable category of the threatened species list under the *Environment Protection and Biodiversity Conservation Act 1999* (Cwth) (EPBC Act) effective from 2 March 2022.

Petaurus australis australis (yellow-bellied glider (south-eastern)) was assessed by the Threatened Species Scientific Committee to be eligible for listing as Vulnerable under Criterion 1. The Committee's assessment is at Attachment A. The Committee's assessment of the subspecies' eligibility against each of the listing criteria is:

- Criterion 1: A4(c): Vulnerable
- Criterion 2: Not eligible

- Criterion 3: Not eligible
- Criterion 4: Not eligible
- Criterion 5: Insufficient data

The main factors that make the yellow-bellied glider (south-eastern) eligible for listing in the Vulnerable category are population reduction and habitat destruction following the 2019–20 bushfires and continuing population decline due to land clearing, fragmentation, extensive severe fires, and climate change.

Species can also be listed as threatened under state and territory legislation. For information on the current listing status of this subspecies under relevant state or territory legislation, see the Species Profile and Threat Database.

Species information

Taxonomy

The species is conventionally accepted as *Petaurus australis* Shaw (1791).

Two subspecies are recognised and accepted but have not yet been formally described. *Petaurus australis australis* (yellow-bellied glider (south-eastern)) comprises most of the distribution and contributes the most to the population size of the subspecies. *Petaurus australis* (Wet Tropics subspecies) (yellow-bellied glider (Wet Tropics)) is an isolated northern subspecies that is recognised as being genetically distinct and is proposed by Brown et al. (2006) as a distinct Evolutionarily Significant Unit (ESU). Preliminary genetic research supports the recognition of separate subspecies (Cooper et al. 2018), and it is suggested that each subspecies is managed separately (Threatened Species Recovery Hub 2020).

Haplotype analysis suggests that, within the south-eastern subspecies, distinct haplotypes may distinguish the south-west Victorian (Vic)/South Australian (SA) subpopulations from the other south-eastern subpopulations (Brown et al. 2006). All western Vic and SA subpopulations share a single haplotype, which suggests low genetic diversity in the region. Given there is no longer a continuous distribution of the subspecies across western Vic, these subpopulations should be regarded as distinct Management units (Threatened Species Recovery Hub 2020). However, the haplotype differences may be a result of isolation by distance and should not preclude translocations to these subpopulations to improve genetic diversity. Further research is required to assess the ESU status of isolated subpopulations in western Vic and SA, and to prioritise subpopulations with high genetic diversity for conservation and low genetic diversity for translocations (Brown et al. 2006; Threatened Species Recovery Hub 2020).

Description

The yellow-bellied glider is a medium-sized arboreal marsupial, the largest Australian petaurid (Russell 1995) and the second largest Australian glider. The head and body length have a range of 240–310 mm, with a 380–470 mm long tail. The body is a greyish-brown colour with a black stripe running down the back and extending to the tail. The tail is mostly black with grey edging at the base. Males weigh 470–725 g and females weigh 435–660 g, though the head and body length of males is only marginally longer than that of females, and females have longer tails. The belly is white to yellow, typically paler in young individuals and becoming more yellow with age. There are black markings on the feet, with a black stripe running down each thigh. There are also black markings along the edge of the gliding membrane. The ears are pale in colour and bare, featuring prominently on the head. This description was drawn from Goldingay (2008).

The two subspecies of yellow-bellied glider are similar in appearance, though the yellow-bellied glider (Wet Tropics) is typically smaller than the yellow-bellied glider (south-eastern) (Goldingay & Kavanagh 1991). The Wet Tropics subspecies may also be also darker in colour on the back than the south-eastern subspecies and possesses less distinctly yellow belly fur (Brown et al. 2006).

Distribution

The yellow-bellied glider (south-eastern) is found at altitudes ranging from sea level to 1400 m above sea level and has a widespread but patchy distribution from south-eastern Queensland (Qld) to far south-eastern SA, near the SA-Vic border (Map 1). In NSW, it predominantly occurs in forests along the eastern coast, from the NSW-Qld border to the NSW-Vic border. However, the distribution also extends inland to the western slopes of the Great Dividing Range in parts of NSW and Qld (van der Ree et al. 2004). The distribution of the yellow-bellied glider (south-eastern) overlaps with the Gondwana Rainforests of Australia World Heritage Area.

In Vic, 75 percent of all yellow-bellied glider (south-eastern) records are in the eastern portion of the state, extending from the east coast to Melbourne and Port Philip bay. The subpopulations in the region of the Vic–SA border are isolated from the main distribution (Carthew 2004; Rees et al. 2007). These isolated western subpopulations comprise 25 percent of Victorian records, and include subpopulations around Edenhope, Portland, Timboon, and the Otway Ranges (Rees et al. 2007). They may represent evolutionarily significant units and be genetically distinct from the other parts of the distribution (Brown et al. 2006). It is recommended these subpopulations be treated as distinct Management Units (Threatened Species Recovery Hub 2020).

As of 2004, it was suggested that there were only six individuals left in SA, all in one 200 ha area of forest (Carthew 2004). Carthew (2004) also suggests that yellow-bellied gliders (south-eastern) were formerly more abundant and widespread in the state. This is based on evidence of distinctive older V-shaped incisions suspected to be made by the subspecies on sap trees in areas it is not currently found. There are no known records of the subspecies in SA since 2010, and it is now potentially extinct from SA (DEW 2021).

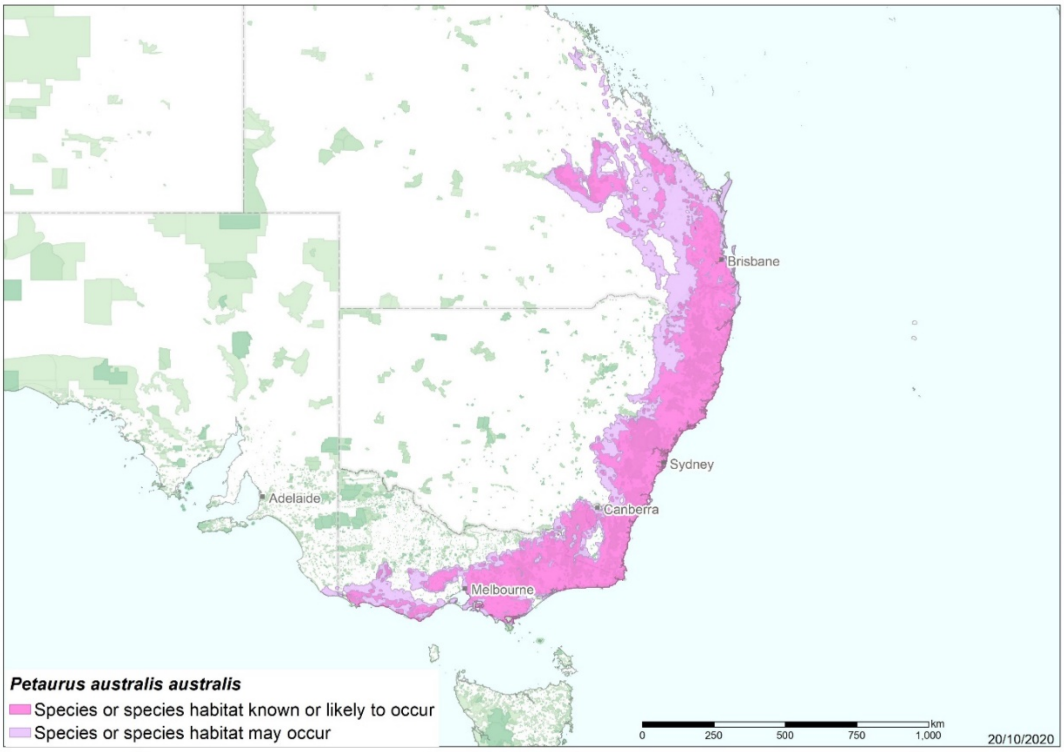
As in NSW, most of the Qld distribution is coastal, extending southward along the eastern seaboard from north of Mackay and continuing through the NSW-Qld border. However, isolated subpopulations are found inland in the Blackdown and Canarvon Ranges of central Qld (Eyre 2004).

Across the entire range, the subspecies' distribution is highly disjunct due to a combination of biogeographic processes and land clearing (Carthew 2004; van der Ree 2004; Rees et al 2007). The specific habitat requirements of the subspecies also lead to disjunct distributions, even in continuous sections of forest (Eyre 2004). Small social groups occupy large and exclusive home ranges and occur at low densities (0.03-0.14 individuals/ha: Henry & Craig 1984 cited in Woinarski et al. 2014; Goldingay & Kavanagh 1993; Goldingay & Jackson 2004; Woinarski et al. 2014).

Monitoring methods

Wire cage traps placed on sap trees have been commonly used for population estimates of the yellow-bellied glider (south-eastern) (Craig 1985; Goldingay & Kavanagh 1990; Goldingay 1992). However recent studies have forgone trapping in favour of a count of night-time calls, often with a playback used to stimulate calling. The yellow-bellied glider is the most vocal of all marsupials. Its loud shrieking calls are audible 500 m away, making auditory sampling suitable for the species. Goldingay et al. (2017) made use of night-time calling, in response to broadcasts of calls of both yellow-bellied glider (south-eastern) and predatory owls. They determined that some spotlighting surveys can overestimate the number of individuals in an area if there is insufficient spacing between survey transects and if too much time is spent within a large area occupied by a single social group (Goldingay et al. 2017). The same individuals can be counted twice if they move between feed trees, thus inflating estimated group size. This can be overcome by conducting shorter spotlight surveys (20 minutes) over a shorter space (200 m) and relying on animals calling in quick succession at different spatial locations to score more than one individual as being present, minimising the chance of movement by individuals causing double scoring (Goldingay et al. 2017). The detection probability per night of surveying for the yellow-bellied glider (south-eastern) is estimated at 0.41 (0.34–0.49) (Wintle et al. 2005), though Goldingay et al. (2017) found a range from 0.28 to 0.71 and notes detection probability varies depending on the season. Southwell et al. (2020) recommends six nights of spotlighting for 0.95 probability of detection using a 10-minute point count with 15 minutes of playbacks followed by 40 minutes of spotlighting.

Map 1 Modelled distribution of the yellow-bellied glider (south-eastern)



Source: Base map Geoscience Australia; species distribution data Species of National Environmental Significance database.
Caveat: The information presented in this map has been provided by a range of groups and agencies. While every effort has been made to ensure accuracy and completeness, no guarantee is given, nor responsibility taken by the Commonwealth for errors or omissions, and the Commonwealth does not accept responsibility in respect of any information or advice given in relation to, or as a consequence of, anything containing herein.
Species distribution mapping: The species distribution mapping categories are indicative only and aim to capture (a) the specific habitat type or geographic feature that represents to recent observed locations of the species (known to occur) or preferred habitat occurring in close proximity to these locations (likely to occur); and (b) the broad environmental envelope or geographic region that encompasses all areas that could provide habitat for the species (may occur). These presence categories are created using an extensive database of species observations records, national and regional-scale environmental data, environmental modelling techniques and documented scientific research.

Cultural and community significance

The cultural significance of the yellow-bellied glider (south-eastern) is not known. It is found across land belonging to many Traditional Owner cultural and dialectal groups. It is likely that the distribution of the subspecies intersects with Aboriginal heritage sites of cultural, scientific, and historic interest.

Relevant Biology/Ecology

Habitat ecology

The yellow-bellied glider (south-eastern) occurs in eucalypt-dominated woodlands and forests, including both wet and dry sclerophyll forests (Kavanagh et al. 1995; Rees et al. 2007). Abundance is highly dependent on habitat suitability, which is in turn determined by forest age and floristics (Woinarski et al. 2014). The subspecies shows a preference for large patches of mature old growth forest that provide suitable trees for foraging and shelter (Milledge et al.

1991; Eyre & Smith 1997; Incoll et al. 2001; Eyre & Goldingay 2003; Eyre 2002, 2004; van der Ree et al. 2004; Kavanagh et al. 2021). There is also a clear preference for forests with a high proportion of winter-flowering and smooth-barked eucalypts (Kavanagh 1987a; Eyre & Smith 1997; Eyre 2004; Irish & Kavanagh 2011; Woinarski et al. 2014). Smooth-barked eucalypts are important due to the range of foraging substrates (and therefore food resources) they provide, as loose bark hanging in strips from these trees provides shelter for insect prey (Eyre & Smith 1997). Yellow-bellied gliders (south-eastern) also require some level of floristic diversity to provide a year-round food supply, and they are unlikely to persist in forests dominated by only one or two tree species (Kavanagh 1987a). Many tree species are found in the subspecies' habitat, with some used for sap feeding. A list of trees known to be used for sap-feeding is provided in Appendix A; note that this list is not comprehensive and local observations should be sought to evaluate habitat use in a given area.

The yellow-bellied glider (south-eastern) is nocturnal. It is active for most of the night and devotes 90 percent of the time spent outside the den to foraging-related activities (Goldingay 2008). The subspecies is social and lives in family groups of two to six individuals (though usually three to four) of varying age and sex composition, throughout an exclusive home range of approximately 50–65 ha (plausible range 25–85 ha) (Craig 1985; Goldingay 1992; Goldingay & Kavanagh 1993; Goldingay & Possingham 1995; Goldingay & Quin 2004). Home ranges are necessarily large, because the trees used as foraging substrates are dispersed and use of trees can vary through time and space (Woinarski et al. 2014). These ranges are defended territories and are advertised by vocalisations (Goldingay 1994; Goldingay et al. 2011). Due to these large home ranges, large areas of forests are required to maintain subpopulation viability. Goldingay and Possingham (1995) suggest that minimum habitat areas of 180–350 km^2 are required to maintain a viable subpopulation, with a minimum of 150 glider groups within a habitat area required to achieve a probability of persistence of 0.95 over 100 years. Eyre (2002) suggests that 320 km^2 of forest is the minimum area required for subpopulation viability in southern Qld. Kambouris et al. (2013) suggests that approximately 350 km^2 of suitable habitat available on the Bago Plateau would support over 200 family groups of the subspecies.

The subspecies has very low dispersal capabilities over spaces larger than its gliding distance. The maximum gliding distance may be up to 120 m–140m (Kavanagh & Rohan-Jones 1982; Kambouris et al. 2013; Goldingay 2014), though management should be informed by average gliding performance (Goldingay 2014). Average performance in low-canopy forest has been documented at 25.2 m, and it is suggested a glide ratio (horizontal distance/height dropped) of 2.0 should be used to estimate gliding distance for management decisions (Goldingay 2014).

During the day, the yellow-bellied glider (south-eastern) shelters in hollows found in large, old trees, usually more than one metre in diameter (Kambouris et al. 2013). Hollow-bearing trees are a critical habitat feature for the yellow-bellied glider (south-eastern) (Goldingay 2011; Goldingay et al. 2019) due to their usage as dens. Indeed, in the montane ash forests of central Vic, the subspecies' distribution is associated positively with the presence of old growth forest (Lindenmayer et al. 1999a) though this old growth ash forest covers just 0.47 to 1.16 percent of the landscape in the Central Highlands of Victoria (Lindenmayer & Taylor 2020).

Hollow-bearing trees used by the yellow-bellied glider (south-eastern) are primarily living, smooth-barked eucalypts of multiple species. Stags (standing dead trees) account for only two percent of den trees in certain forest types (Goldingay 2011). Trees do not usually start producing hollows until they reach at least 50 cm diameter at chest height (Goldingay 2011). Age is critical in the formation of hollows, as trees above 50 cm in diameter at chest height are usually above 100 years of age (Mackowski 1984; Wormington & Lamb 1999; Wormington et al. 2003; Koch et al. 2008). This age requirement makes management of species relying on hollow-bearing trees difficult, as it is hard to maintain a supply of hollows when existing trees are lost to disturbances (Ball et al. 1999). The preference of the yellow-bellied glider (south-eastern) for living hollow-bearing trees may mean that the death of hollow-bearing trees through fire is particularly harmful to the subspecies. However, Eyre & Smith (1997) reported that yellow-bellied glider (south-eastern) abundance increased significantly as the number of dead hollow-bearing trees increased, though noted this result was not expected and the subspecies generally dens in live trees. This was attributed to competition with other hollow-dependent gliders, with scarcity of living hollow-bearing trees potentially forcing the occupancy of dead trees.

Diet

Sap drawn from excisions in sap trees forms an important component of the diet of the yellow-bellied glider (south-eastern), especially when alternative food sources are limited (Goldingay 1991; Eyre & Goldingay 2005). The animals use their large lower incisor teeth to make characteristic V-shaped cuts and vertical furrows in the tree trunk (Goldingay 2008; Goldingay & Kavanagh 1991). Yellow-bellied gliders (south-eastern) also feed on insects, spiders, eucalypt nectar and pollen, insect exudates and manna (Carthew et al. 1999; Eyre & Goldingay 2003; Goldingay 2008).

Sap feed trees are a critical habitat feature for the yellow-bellied glider (south-eastern). A variety of species are used as sap trees, though it appears that fewer than 10 feed trees are utilised by an individual or family group (Goldingay & Kavanagh 1991) and tree species used at one site are not always used at other sites (Kavanagh 1987a, b; Goldingay 1991). In NSW, 37 species have been identified as sap trees (NPWS 2003; Appendix A). Most are eucalypts, though use of other taxa has also been documented. A study based in southern Qld identified 13 species used as sap trees, with the species *Eucalyptus longirostrata* (Grey Gum) and *Eucalyptus biturbinata* (Grey Gum) most likely to be exploited (Eyre & Goldingay 2005). The factors determining which trees are used as sap trees are not fully understood, though one hypothesis suggests gliders respond to an intermittent increase in sap flow found only in certain individual trees (Mackowski 1988; Eyre & Goldingay 2005; Goldingay 2008).

Reproductive ecology

The subspecies reproduces seasonally, with timing varying across its broad range (Woinarski et al. 2014). Litter size is usually one, and individuals reach sexual maturity at around two years of age (Woinarski et al. 2014). Once sexual maturity is reached, individuals will pair up with each other, usually in a monogamous relationship. The subspecies' lifespan is a minimum of six years in the wild. Generation length is four to five years (Goldingay & Kavanagh 1991; Woinarski 2014).

Habitat critical to the survival

Key considerations in environmental impact assessments relevant to the yellow-bellied glider (south-eastern) should include the following factors about the subspecies:

- Highly specialised diet and requirement of trees with unknown characteristics for sap feeding;
- Prefer to den in living hollow-bearing trees;
- Groups live in large, exclusive home ranges as foraging substrates are dispersed and variable through time and space; and
- Large areas of forest are required to maintain population viability.

Habitat critical to the survival of the yellow-bellied glider (south-eastern) may be broadly defined as areas containing the following attributes (noting that geographic areas containing habitat critical to survival needs to be defined by forest type on a regional basis):

- large contiguous areas of floristically diverse eucalypt forest, which are dominated by winter-flowering and smooth-barked eucalypts, including mature living hollow-bearing trees and sap trees (see *Appendix A*);
- areas identified as refuges under future climate change scenarios;
- short or long-term post-fire refuges (i.e., unburnt habitat within or adjacent to recently burnt landscapes) that allow the species to persist, recover and recolonise burnt areas;
- habitat corridors required to facilitate dispersal of the subspecies between fragmented habitat patches and/or that enable recolonization or movement away from threats. yellow-bellied gliders (south-eastern) have a glide ratio (horizontal distance/height dropped) of around 2.0, and corridors spanning gaps larger than the distance gliders are likely to be able to travel should be considered critical to the survival. There is not enough evidence to define the canopy and width characteristics of appropriate corridors. In the absence of such information, a precautionary approach should be taken to maximise dispersal by considering all habitat corridors in the species' range to be habitat critical to the survival; and
- areas in which some trees have evidence of use for sap extraction by yellow-bellied glider (south-eastern).

Note: In addition to the above, other attributes associated with a particular area can help evaluate its value and role in a species' life cycle and recovery – e.g., the frequency of use of that area, the area's ability to become habitat for the species, the area's ability to provide habitat during times of stress, or the area's ability and cost-effectiveness of it to be managed for the species so that the species can be re-established in that area.

Habitat meeting any one of the criteria above is considered habitat critical to the survival of the yellow-bellied glider (south-eastern), irrespective of the abundance or density of the species or the perceived quality of the site. Forest areas currently unoccupied by the yellow-bellied glider (south-eastern) may still represent habitat critical to survival, if the recruitment of hollow-bearing trees in the future could allow the species to colonise these areas and ensure persistence of a population. Whenever possible, habitat critical to the survival of the species should not be destroyed or modified. Actions that have indirect impacts on habitat critical to survival should be minimised (e.g. clearing, road construction), and actions that compromise adult and juvenile survival should also be avoided.

No Critical Habitat as defined under section 207A of the EPBC Act has been identified or included in the Register of Critical Habitat.

Important populations

In this section, the word population is used to refer to subpopulation, in keeping with the terminology used in the EPBC Act and state/territory environmental legislation.

Populations important to the survival of the yellow-bellied glider (south-eastern) include stronghold populations, ecologically or genetically distinct populations (e.g., those at the limits of the subspecies' range, outlying populations), research populations, and other populations where recovery actions are being implemented. However, this list of important populations is not exhaustive. In the absence of such information, all known populations should be considered important. Known important populations include:

- Carnarvon Range (Inland population; Qld)
- Blackdown Tableland (Inland population; Qld)
- Bago Plateau (Endangered under NSW legislation; NSW)
- Richmond Range National Park (research population; NSW)
- Blacktown range (population near urban area; NSW)
- Shoalhaven populations (severely fire-affected, surveyed; NSW)
- Populations between Coffs Harbour, Dorrigo, Glen Innes and Grafton (affected by fire and timber harvesting, research populations; NSW)
- Populations between Nimmitabel and Cathcart (affected by fire and timber harvesting, research populations; NSW)
- Populations near Waratah Creek (affected by fire and timber harvesting, research populations; NSW)
- South Australian population (only SA population, potentially an ESU, six individuals, may be extinct; SA)
- Western Vic populations (outlying populations, potentially an ESU; Vic).

Threats

The yellow-bellied glider (south-eastern) is primarily threatened by climate change, altered fire regimes, clearing, fragmentation and timber harvesting. Invasive species have been recorded preying on the subspecies, though it is unknown if this is having a population-level impact. Other minor threats include mortality by barbed wire fencing and habitat degradation by feral deer (Cervidae spp.). Dieback caused by *Phytophthora cinnamomi* may also be impacting the subspecies through habitat degradation, though more research is required to determine whether the effect of dieback is impacting the subspecies at a population level.

Table 1 Threats impacting yellow-bellied glider (south-eastern)

Threat	Status and severity [a]	Evidence
Habitat loss, disturbance and modification		
Habitat clearing and fragmentation	• Status: current • Confidence: known • Consequence: major • Trend: decreasing • Extent: across parts of the range	Much of the forest in south-eastern Australia is fragmented due to extensive land clearing for development and agriculture. This has likely impacted yellow-bellied glider (south-eastern) abundance, as the subspecies and other arboreal marsupials are known to be particularly susceptible to the impacts of clearing (Lindenmayer et al. 1999a; 1999b; 1999c; Youngentob et al. 2013). In NSW, the area occupied by the yellow-bellied glider (south-eastern) is thought to have declined by 26-50% since European settlement (Lunney et al. 2000, cited in DPIE 2014), largely due to habitat loss, fragmentation and other landscape-scale effects (e.g., fire). Taylor et al. (2011) found evidence that fragmentation may result in isolation in the genetic structure of *Petaurus norfolcensis* (squirrel glider), a congeneric of the yellow-bellied glider (south-eastern) that also relies on hollows and sap feeding, and Goldingay et al. (2013) found for the same species that loss of habitat connectivity reduced genetic diversity in isolated populations over small spatial scales (1-50 km) and short time periods (20-50 years). The yellow-bellied glider (south-eastern) is particularly vulnerable to the impacts of clearing and fragmentation due to its large, exclusive home ranges, requirement for large areas of forest, and inability to cross even small areas of cleared land (Kambouris et al. 2013; Woinarski et al. 2014). It may not persist in small, isolated forest fragments (Lindenmayer 1999, cited in Taylor & Rohweder 2020), and barriers such as roads potentially reduce gene flow and cause genetic drift, as well as increase the risk of localised extinction through environmental and demographic events (NPWS 2003). Loss of connectivity may be managed using gliding poles, which yellow-bellied gliders (south-eastern) have been observed to use (Goldingay et al. 2019; Taylor & Rohweder 2020). Gliding poles enable gliding mammals to cross wide road corridors and other gaps, restoring functional connectivity to habitat fragmented by major roads. yellow-bellied gliders (south-eastern). Much of the current habitat for the yellow-bellied glider (south-eastern) is found in conservation reserves across its range (Woinarski et al. 2014), which may help to constrain the future impact of clearing on the subspecies.
Extensive severe bushfires	• Status: current • Confidence: known	Projections of higher temperatures and reduced mean rainfall for eastern Australia due to climate change are leading to increased frequency and severity of bushfires (CSIRO & Bureau of Meteorology 2015). This is most

Threat	Status and severity [a]	Evidence
	• Consequence: major • Trend: increasing • Extent: across the entire range	clearly evidenced by the catastrophic bushfires of 2019–20, where an unusually large area burned at high severity, (DPI 2020) intersecting with 41 percent of the distribution of the yellow-bellied glider (Legge et al. 2021). If bushfires continue to recur at a similar scale and severity, this will likely drive substantial population reductions and local extinctions of yellow-bellied glider (south-eastern) (Woinarski et al. 2014). Direct mortality due to fire occurs through lethal heating or sublethal inhalation of smoke, and indirect mortality occurs due to the loss of important habitat features and resources such as sap trees and live hollow-bearing trees (Lunney 1987; Goldingay & Kavanagh 1991; Bradstock et al. 2005). Previous high severity fires have eliminated yellow-bellied gliders (south-eastern) from a site for at least 15 years (Goldingay & Kavanagh 1991), and local extinctions due to high severity fire have been observed (D Lindenmayer pers comm, cited in Woinarski et al. 2014). McLean (2012) found that the yellow-bellied glider (south-eastern) was mostly absent from sites that had experienced fire c. 10 years prior to sampling, and it was more common on long unburnt sites. Eyre (2005) found that the abundance of dead hollow-bearing trees was significantly impacted by fire severity, and Goldingay (2021) found a negative relationship between dead tree abundance and occupancy. It is considered highly likely that the subspecies is negatively influenced by fire severity (Goldingay 2021). Site-level population declines from the 2019–20 bushfires are estimated at 82 percent for severely affected sites using expert elicitation, and post-fire on-ground surveys suggests that declines may be up to 83–97% (see *Criterion 1*). In areas where direct mortality is high, maintenance of subpopulations is dependent on rates of dispersal and recolonization (Bradstock et al. 2005). Given that the yellow-bellied glider (south-eastern) has very low dispersal (Goldingay & Kavanagh 1990, 1991), the ability of the subspecies to recolonise unburnt areas is likely very low. Since European settlement, fire regimes in Australia have been significantly altered by a combination of land use changes and clearing for development and agriculture (Cresswell & Murphy 2016). Changes to the existing fire regime include a shift to very large fires occurring at shorter intervals, which is a threat to forest-dwelling mammals in south and south-eastern Australia (Lindenmayer 2015). Though yellow-bellied glider (south-eastern) subpopulations may remain stable under natural fire regimes, an increase in the severity, size and frequency of fires is likely to have a large impact on the viability of many subpopulations.
Prescribed burns	• Status: current • Confidence: known • Consequence: moderate • Trend: increasing • Extent: across parts of the range	In 2016, DELWP investigated the impact of prescribed burning on hollow-bearing trees, finding that trees burnt by planned burns were 28 times more likely to collapse than unburnt trees (Bluff 2016). This high collapse rate results in the loss of habitat for hollow-dependent fauna such as the yellow-bellied glider (south-eastern). DELWP suggests the most effective method to manage this threat is to cease burning of areas known to be inhabited by hollow-dependent fauna and conduct mechanical fuel reduction instead. Loss of hollow-bearing trees due to site preparation works associated with prescribed burns may also affect habitat for the yellow-bellied glider (south-

Threat	Status and severity [a]	Evidence
		eastern). This has been documented in foothill forests close to settled areas in Vic, though further study is required to determine the specific outcomes of prescribed burning on biodiversity (Bluff 2016). Since 2019, the reliance on controlled fire for regeneration in Victoria has decreased and there has been a shift away from high severity burning to low/moderate severity regeneration burns and mechanical disturbance (VicForests 2021. pers comm 24 June). McLean (2012) suggests that the use of prescribed burning poses a management paradox for the subspecies and found that trees with fire-induced injuries around the base are more likely to have hollows (McLean 2012). Therefore, low severity fires without canopy scorch may help provide this critical habitat feature for the subspecies in the longer term (McLean 2012). However, such fire regimes will also cause a shorter-term decline in the abundance of arboreal mammals due to direct impacts (McLean 2012). Further research is needed to determine whether a management paradox exists, as the suitability of the additional tree hollows provided by fire for use by fauna is relatively unknown and substantial losses of hollow-bearing trees have been measured in a range of forest ecosystems following fire (Inions et al. 1989; Parnaby et al. 2010; Bluff 2016).
Timber harvesting	• Status: current • Confidence: known • Consequence: major • Trend: decreasing • Extent: across parts of the range	Timber harvesting is a key threat to arboreal marsupials (Eyre & Smith 1997; Lindenmayer et al. 2020) through the depletion of large mature trees (McLean et al. 2015), and loss of canopy connectivity and denning/food resources that these provide (Kavanagh 1987; Kavanagh & Lambert 1990). The yellow-bellied glider (south-eastern) prefers old growth forest containing a high abundance of hollow-bearing trees, in comparison to regrowth forest or forest subject to selective logging rotations (Lindenmayer et al. 1999a; Incoll et al. 2001; Alexander et al. 2002; McAlpine & Eyre 2002). Though the subspecies has been observed using nest boxes (Goldingay et al. 2020), more research is required to determine the usefulness of these structures in mitigating hollow-bearing tree loss. In the Grafton/Casino Forestry Management Area (FMA) of NSW, the subspecies was not recorded during surveys of recently logged forests (Smith et al. 1994), and in the Urbenville FMA of NSW, it was found more frequently in unlogged mature and old growth forests compared to logged forests (Andrews et al. 1994). Similarly, in the Mountain Ash forests of Vic, Milledge et al. (1991) also reported that the subspecies was significantly more abundant in old growth forest, and south-east NSW subpopulations had not recovered after eight years at sites where timber harvesting removed 62%, 52% and 21% of the basal tree area (Kavanagh & Webb 1998). Though density did not significantly change in this study, the detectability of the subspecies in logged areas decreased, counts were consistently greater on the unlogged control, and logged areas were used rarely (Kavanagh & Webb 1998). In Qld, Eyre (2007) also found that the occurrence of the yellow-bellied glider (south-eastern) was negatively correlated with timber harvesting, though clearing within 1 km of habitat had a much greater influence on site occupancy than timber harvesting.

Threat	Status and severity [a]	Evidence
		The large home ranges of the yellow-bellied glider (south-eastern) make them less sensitive to timber harvesting compared to similar arboreal species, such as *Petauroides volans* (greater glider). The subspecies can persist in areas impacted by timber harvesting, provided that old trees are preserved within riparian zones and unharvested forest patches and corridors are retained adjacent to logged land (Goldingay & Kavanagh 1991; Goldingay & Kavanagh 1993; Alexander et al. 2002; Eyre & Smith, 1997; Goldingay & Kavanagh, 1991; Kavanagh & Bamkin 1995; Kavanagh & Webb, 1998). The preservation of riparian strips, habitat patches, connectivity corridors and some hollow-bearing trees during timber harvesting has been a practice over the past few decades in Vic (VicForests 2021. pers comm 24 June), though areas harvested prior to this were likely clearfelled. It is likely that the impact of timber harvesting on the yellow-bellied glider (south-eastern) is dependent on the intensity of the harvesting process, the forestry practices used, and the forest type. Occurrence of the subspecies has been found to decrease with increased logging intensity (Smith et al. 1994; Smith et al. 1995, cited in DPIE 2014; Wormington et al. 2002; Eyre 2007), and only minor differences in abundance due to timber harvesting have been observed when the complexity of the forest mosaic is high (i.e., when unlogged forest was always next to logged forest) (Lunney 1987; Kavanagh & Bamkin 1995). These and other studies also found no significant association between harvesting history and yellow-bellied glider (south-eastern) presence and density (Kambouris et al. 2013; Kavanagh & Bamkin, 1995; Lunney, 1987; McLean, 2012), though Kavanagh & Bamkin (1995) found twice as many individuals in unharvested areas. The impact of forestry may be more pronounced in habitat such as Mountain Ash forests, where the complexity and diversity of floristics is lower than at the sites studied in NSW (Lunney; 1987; Kavanagh & Bamkin 1995; Kavanagh & Webb, 1998; Kambouris et al. 2013). Areas which are of high conservation value for threatened forest-dependent species in Vic may be targeted for future timber harvesting under the 2019 Timber Release Plan (Taylor & Lindenmayer 2019). However, a subsequent plan has been published (VicForests 2020), and the impacts of timber harvesting may lessen across the Vic distribution due to the implementation of new harvesting techniques and the scheduled cessation of native forest timber harvesting in 2030 (VicForests 2021. pers comm 24 June). An analysis by VicForests suggests that just 1.9% of the high-quality habitat for the yellow-bellied glider (south-eastern) in Vic may feasibly be impacted by timber harvesting between 2021 and 2030, not including retained strips and patches of riparian forest (VicForests 2021. pers comm 24 June). However, it is unclear how much habitat will be impacted that is not considered 'high quality', and how the quality of habitat was determined. A Habitat Distribution Model for the yellow-bellied glider (south-eastern) in Vic indicates that around 62% of its habitat is in multiple use zones of State Forest, which may be subject to logging, and 38% is in National Park and other formal reserves. However, some species-specific and habitat value protections are in place, and yellow-bellied gliders (south-eastern) will also benefit from protections to other species. For example, a further 6% of

Threat	Status and severity [a]	Evidence
		its habitat has been reserved via Greater Glider Immediate Protection Areas (IPAs), from which harvesting is excluded (VicForests 2021. Pers comm 24 June). The habitat overlap of the subspecies with State Forests in NSW and Qld is unclear. Interactions of timber harvesting with fire Timber harvesting can interact with fire to compound the impact of bushfires on hollow-bearing trees. McLean (2012) found that following fire, logged sites exhibited a higher loss of hollow-bearing trees than unlogged sites (although logging in this study commenced during the 1960s, prior to modern timber harvesting operations such as variable retention and selective harvesting systems). After the 2009 Victorian bushfires, 79% of live hollow-bearing trees died at a study site in the Mountain Ash forests, with no subsequent recruitment of hollow-bearing trees in 2011 (Lindenmayer et al. 2012). This was attributed to past bushfires and timber harvesting. Timber harvesting regimes in eucalypt forests may make those forests more likely to experience crown fire (fire that leaps between canopies) (Lindenmayer et al. 2009; Price & Bradstock 2012; Taylor et al. 2014). Timber harvesting can alter key attributes of forests by changing microclimates, stand structure and species composition, fuel characteristics, the prevalence of ignition points, and patterns of landscape cover. Such changes may make some forest types more prone to ignition and increased fire severity (Lindenmayer et al. 2009). Other studies, including one after the 2019–20 bushfires, disagree with this conclusion, and suggest logging has no impact on fire risk and severity (Attiwill et al. 2013; Bowman et al. 2021). However, this is contested by other researchers (Zylstra et al. 2021). Hollow-bearing trees are also threatened by post-fire salvage timber harvesting (Noss & Lindenmayer 2006; Lindenmayer et al. 2008). In Vic, adaptive management measures have been implemented after the 2019–20 fires to curb habitat loss, including identifying and reserving suitable habitat around harvesting coupes within the bushfire footprint and undertaking pre-harvest surveys in high quality habitat (VicForests 2021. pers comm 24 June). The NSW EPA has also issued the Forestry Corporation of NSW with site-specific conditions for harvesting at fire-affected sites, including additional protection of hollow-bearing and feed trees (EPA 2021), and timber harvesting has resumed in some NSW state forests (Forestry Corporation 2021).
Climate change		
Increased temperatures and changes to precipitation patterns	• Status: current/future • Confidence: known • Consequence: catastrophic • Trend: increasing • Extent: across the entire range	Projections of higher temperatures and reduced mean rainfall for eastern Australia due to climate change are leading to increased frequency and severity of droughts and bushfires (CSIRO & Bureau of Meteorology 2015). This is most clearly evidenced by the droughts occurring from 1997–2009 and 2017–2019 (DPI 2020; BOM 2021) and subsequent catastrophic bushfires of 2019–20, where an unusually large area burned at high severity (DPI 2020). The yellow-bellied glider (south-eastern) may be vulnerable to the combination of these threats, as drought conditions can act in tandem with bushfires to reduce the

Threat	Status and severity [a]	Evidence
		abundance of small and medium-sized marsupials (Hale et al. 2016; Crowther et al. 2018). Gradual declines in subpopulations of the subspecies in north-eastern NSW may have occurred due to climate change, though this requires further investigation (Kavanagh et al. 2021). During surveys in north-eastern NSW, only 25 percent of previously utilized sites were occupied and there was no evidence of recent sap tree usage (R Kavanagh 2021. pers comm 10 August). The driving factors behind these potential declines are unclear, though climate change may be responsible through reductions to rainfall and increases in mean daytime and night-time temperatures (R Kavanagh 2021. pers comm 10 August). The related greater glider is sensitive to extreme temperatures (Rübsamen et al. 1984) and is likely to undergo severe range contractions as temperatures increase (Kearney et al. 2010). Indeed, higher night-time temperatures were considered to be the cause of declines of greater gliders at multiple sites (Smith & Smith 2018, 2020; Wagner et al. 2020). Elevated temperatures, combined with drought stress may lead to similar declines in yellow-bellied glider (South-eastern). It is also possible that such conditions alter the sap flow of sap feeding trees, but this has not been investigated. Further research is required to determine the impact of climate change on population decline (R Kavanagh 2021. pers comm 10 August). Rising temperatures will result in elevated water requirements for arboreal marsupials and in some cases water stress, forcing range contraction into climate refugia (Kearney et al. 2010; Krockenberger et al. 2012). Indeed, climate change is likely to influence the distribution of the yellow-bellied glider (south-eastern) through a shift of bioclimates that support the subspecies' habitat (Handayani et al. 2019). A study modelling yellow-bellied glider (south-eastern) habitat loss due to climate change in south-east Qld found that there will likely be substantial decreases in core and marginal habitat for the subspecies, even under low warming scenarios. Contraction of suitable habitat may not be linear with emissions and time, as core habitat may expand in the south-west of NSW as it contracts in Qld. In general, the reduction is predicted to occur mostly in Qld, with only small pockets of habitat remaining in the state (Handayani et al. 2019).
Introduced species		
Predation by European red foxes (*Vulpes vulpes*)	• Status: current • Confidence: suspected • Consequence: minor • Trend: unknown • Extent: across the entire range	Yellow-bellied gliders (south-eastern) have been found in the scats of European red foxes (Lunney et al. 1990; Mitchell & Banks 2005). Previously, it was thought that these predators cannot climb into the canopy where gliders are found, so it was assumed they were eating already dead animals. However, video evidence from 2017 shows that foxes can and do climb trees (Mella et al. 2017), meaning that some predation on living gliders may occur, though this is unlikely to have a population-level impact.
Predation by feral cats (*Felis catus*)	• Status: current • Confidence: suspected • Consequence: minor	There are no known records of feral cats eating yellow-bellied gliders (Woolley et al. 2019). Analysis of scats has found feral cats have consumed a similar species, the greater glider (Jones & Coman 1981), though it is likely that this was through scavenging rather than predation. There is photographic evidence of feral cats climbing into

Threat	Status and severity [a]	Evidence
	• Trend: unknown • Extent: across parts of the range	yellow-bellied glider (Wet Tropics) sap feeding stations (J. Winter pers comm, cited in DAWE 2020).
Habitat degradation from feral deer	• Status: current • Confidence: suspected • Consequence: not significant • Trend: increasing • Extent: across parts of the range	Feral deer are a threat to the saplings of habitat trees. Where deer density is high, saplings are often destroyed by rubbing, trampling, and grazing (DSEWPC 2011).
Fencing of agricultural land		
Barbed wire fencing (entanglement)	• Status: current • Confidence: known • Consequence: minor • Trend: unknown • Extent: across parts of the range	Occasional losses of individuals have been known to occur due to entanglement in barbed wire fencing (Van der Ree et al. 1999).

Each threat has been described in Table 1 in terms of the extent that it is operating on the subspecies. The risk matrix (Table 2) provides a visual depiction of the level of risk being imposed by a threat and supports the prioritisation of subsequent management and conservation actions. In preparing a risk matrix, several factors have been taken into consideration, they are: the life stage they affect; the duration of the impact; and the efficacy of current management regimes, assuming that management will continue to be applied appropriately. The risk matrix and ranking of threats has been developed in consultation with in-house expertise using available literature.

Table 2 Yellow-bellied glider (south-eastern) risk matrix

Likelihood	**Consequences**				
	Not significant	**Minor**	**Moderate**	**Major**	**Catastrophic**
Almost certain	Low risk	Moderate risk	Very high risk **Prescribed burns**	Very high risk **Timber harvesting** **Extensive severe bushfires** **Habitat clearing and fragmentation**	Very high risk **Increased temperatures and changes to precipitation patterns**
Likely	Low risk **Habitat degradation from feral deer**	Moderate risk	High risk	Very high risk	Very high risk

Likelihood	Consequences				
	Not significant	**Minor**	**Moderate**	**Major**	**Catastrophic**
Possible	Low risk	Moderate risk	High risk	Very high risk	Very high risk
Unlikely	Low risk	Low risk **Barbed wire fencing (entanglement)** **Predation by European red foxes** **Predation by feral cats**	Moderate risk	High risk	Very high risk
Unknown	Low risk	Low risk	Moderate risk	High risk	Very high risk

Categories for likelihood are defined as follows:

Almost certain – expected to occur every year

Likely – expected to occur at least once every five years

Possible – might occur at some time

Unlikely – such events are known to have occurred on a worldwide bases but only a few times

Unknown – currently unknown how often the incident will occur

Categories for consequences are defined as follows:

Not significant – no long-term effect on individuals or populations

Minor – individuals are adversely affected but no effect at population level

Moderate – population recovery stalls or reduces

Major – population decreases

Catastrophic – population extirpation/extinction

Priority actions have then been developed to manage the threat particularly where the risk was deemed to be 'very high' or 'high'. For those threats with an unknown or low risk outcome it may be more appropriate to identify further research or maintain a watching brief.

Conservation and recovery actions

Primary conservation outcome

Population size has stabilised because sufficient areas of habitat are protected from extensive severe fire, fragmentation and timber harvesting, known threats are mitigated and key habitat features (e.g., sap trees, hollow-bearing trees) and habitat connectivity are retained.

Conservation and management priorities

Habitat loss, disturbance and modifications (including fire)

- Ensure suitable habitat is maintained and protected from known threats (such as high severity bushfires) around important subpopulations, as well as in areas where subpopulations have already declined through loss of habitat. When protecting an area, retain sufficient suitable habitat for subpopulation viability.
- Avoid planned burns, clearing, timber harvesting or other disturbance in a 65 ha zone around habitat which has been burnt in the past 10 years.
- Re-assess and revise current prescriptions used for prescribed burning to ensure that the frequency and severity of all fires are minimised, to mitigate the risk of further population decline.
- Maintain current effective prescriptions in production forests where the yellow-bellied glider (south-eastern) is found to support subpopulations of the species, and, if necessary, establish new prescriptions. This includes but is not limited to: appropriate levels of timber harvesting exclusion and timber harvesting rotation cycles, maintenance of wildlife corridors between logged patches, active protection of existing sap trees and hollow-bearing trees from known threats, adequate recruitment of hollow-bearing trees, and maintained use of variable retention systems and selective harvesting systems designed to protect hollow-bearing trees.
- Construct artificial hollows in areas of low hollow availability, utilising appropriate nest boxes and/or chainsaw hollows, the latter of which have the potential to restore degraded habitat and may be used more by arboreal marsupials than nest boxes (Terry et al. 2021).
- Restore connectivity in subpopulations fragmented by major roads, including the use of artificial structures such as rope bridges and glide poles.
- If possible, avoid the use of barbed wire and modify existing barbed wire by replacing the top strand of barbed wire with plain or plastic-coated wire (Clancy & Land for Wildlife 2011).

Climate change

- Protect all habitat projected to be suitable as refuge sites under future climate change scenarios (as informed by research) and maintain and establish connectivity to facilitate movement where possible.

Invasive species (including threats from grazing, trampling, predation)

- If required, implement control measures for introduced predators, including the European red fox and feral cat, in areas burnt by bushfires. The control of introduced herbivores (e.g., feral deer) may also aid habitat recovery.

Stakeholder Engagement

- Seek stakeholder input into assessment and planning processes that include protections for the yellow-bellied glider (south-eastern) and its habitat. This may include environmental impact assessments, park management plans, water resource plans, fire management plans and transport development plans.
- Liaise with private land holders, Traditional Owners, conservation, and land management groups to create guidelines for on-ground management of the yellow-bellied glider (south-eastern). Encourage their engagement in surveying and monitoring of the subspecies.
- Encourage landholders to enter land management agreements, particularly in-perpetuity covenants, that promote the protection and maintenance of private lands with high value for the subspecies.
- Foster public interest in the subspecies and its ongoing conservation.

Survey and Monitoring priorities

- Conduct on-ground surveys to establish habitat and population loss as a result of the 2019–20 bushfires and to provide a baseline for future population monitoring, using methods outlined in the *Monitoring methods* section. Ensure that sufficient spotlighting is completed, and that there is at least a 0.95 probability of detection if the subspecies is present.
- Implement an integrated monitoring program across major subpopulations, linked to the assessment of the effectiveness of management actions. Surveys should include long-term monitoring of known subpopulations to determine trends in abundance and to ascertain their status and viability, as well the trends in the overall population size. If necessary, establish new monitoring programs.
- Monitor the abundance, age and size structure of sap trees and hollow-bearing trees and their responses to management measures. This includes before and after prescribed burns, and before and after timber harvesting.
- Monitor the incidence and impacts of bushfire and timber harvesting in the subspecies range, particularly in areas adjacent to those burnt in the 2019–20 bushfires.
- Map habitat critical to the subspecies' survival (as described earlier in this document).

Information and Research priorities

- Investigate the numbers and densities of mature hollow-bearing trees and sap trees required for subpopulation viability and eco-physiological factors that determine which trees are sap trees.
- Investigate effects of management practices in production forests on persistence, abundance and population viability of the subspecies. Increase understanding of how variable retention,

harvesting rotation cycles, maintenance of wildlife corridors, sap tree and hollow bearing tree protection through selective harvesting systems alter the impacts of timber harvesting on yellow-bellied gliders (south-eastern) and similar species.

- Identify and map subpopulations that should be prioritised for conservation and recovery actions, including important populations (as described earlier in this document) and those which have undergone or are projected to undergo substantial decline. Ensure information is made available to conservation planners and managers.
- Assess the genetic structure and ESU status of the western Vic and SA subpopulations and the value of treating these subpopulations as distinct management units.
- Investigate the driving factors behind non-fire related declines observed in some parts of the range (e.g., sites in north-eastern NSW; Kavanagh et al. 2021), and the potential for widespread impacts through climate change.
- Assess the impacts of fire management and different fire regimes on habitat, subpopulation size, hollow availability, and sap tree availability.
- Assess the impacts of Bell Miner associated dieback on yellow-bellied glider (south-eastern) site occupancy.
- Assess the impacts of localised habitat fragmentation (e.g., through peri-urban expansion and road networks) on the subspecies.
- Investigate the subspecies' response to the use of artificial structures (e.g., artificial hollows, chainsaw hollows, hollows formed by silviculture, rope bridges and glide poles over transport routes), and if necessary, trial different designs to increase usage by the subspecies.
- Undertake habitat suitability modelling under future climate change scenarios and identify areas suitable as climate refuges. This should include modelling of the health of significant habitat species, eco-physiological controls on sap flow, and future distribution of high fire risk and extreme events (especially drought and bushfires).
- Investigate the potential to utilise active management, including different forms of silviculture, to accelerate hollow development, manage fuel loads to mitigate fire risk in production forests.
- Assess the level of threat posed by feral animals, including the European red fox, feral cat and feral deer.

Recovery plan decision

A decision has been made to have a Recovery Plan, as a high level of coordination is required to implement conservation and recovery actions, as well as a high level of support by stakeholders and adaptive management

Links to relevant implementation documents

NSW Saving Our Species Targeted Recovery Plan for *Petaurus australis* (yellow-bellied glider) (2020)

NSW National Parks and Wildlife Service Recovery Plan for the Yellow-bellied glider (*Petaurus australis*) (2003)

[Threat abatement plan for predation by feral cats (2015)]

[Threat abatement plan for predation by the European red fox (2008)]

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The Threatened Species Scientific Committee finalised this assessment on 9 September 2021.

Attachment A: Listing Assessment for *Petaurus australis australis*

Reason for assessment

This assessment follows prioritisation of a nomination from the TSSC.

Assessment of eligibility for listing

This assessment uses the criteria set out in the EPBC Regulations. The thresholds used correspond with those in the IUCN Red List criteria except where noted in criterion 4, sub-criterion D2. The IUCN criteria are used by Australian jurisdictions to achieve consistent listing assessments through the Common Assessment Method (CAM).

Key assessment parameters

Table 3 includes the key assessment parameters used in the assessment of eligibility for listing against the criteria.

Table 3 Key assessment parameters

Metric	Estimate used in the assessment	Minimum plausible value	Maximum plausible value	Justification
Number of mature individuals	<100,000	>10,000	100,000	There is no reliable estimate of the population size of the yellow-bellied glider (south-eastern). Woinarski et al. (2014) suggest that there are over 100 000 mature individuals. However, due to ongoing decline and the 2019–20 bushfires, the current population size is likely to be below 100 000 mature individuals but still substantially above 10 000 mature individuals.
Trend	declining			Across the broad range of the yellow-bellied glider (south-eastern), trends of population decline, and the rate of this decline, cannot be reliably estimated. Past decline over three generations may perhaps approach 30 percent, as suggested in Woinarski et al. (2014). One year after the 2019–20 bushfires, an overall population decline of around 21% (or up to 29%, the lower 80% confidence bound) is suspected under current management. This is expected to increase to 25% (or up to 38%) in three generations after the fires (Legge et al. 2021).
Generation time (years)	4–5	4	5	The generation time of the subspecies is considered to be four–five years (Woinarski et al. 2014).

Metric	Estimate used in the assessment	Minimum plausible value	Maximum plausible value	Justification
Extent of occurrence	1,285,082 km^2	712,991 km^2	unknown	The extent of occurrence (EOO) is estimated at 1,285,082 km^2. This figure is based on the mapping of point records from a 20-year period (2000–2020) obtained from state governments, museums and CSIRO. The EOO was calculated using a minimum convex hull, based on the IUCN Red List Guidelines (IUCN 2019). Woinarski et al. (2014) estimated the EOO as 712,991 km^2, calculated using records from 1992–2012.
Trend	Contracting			The EOO has declined since European settlement due to habitat loss induced by land clearing, fragmentation and fire. This includes the local extinction of some subpopulations (Woinarski et al. 2014). EOO is likely to continue contracting due to loss of suitable habitat resulting from the 2019–20 bushfires, planned burning, land clearing and timber harvesting. See Table 1 for further information.
Area of Occupancy	12,724 km^2	Unknown	14,152 km^2	The AOO is estimated at 12,724 km^2. This figure is based on the mapping of point records from a 20-year period (2000–2020) obtained from state governments, museums and CSIRO. The AOO was calculated using a 2x2 km grid cell method, based on the IUCN Red List Guidelines (IUCN 2019). Woinarski et al. (2014) estimated the AOO as 14,152 km^2, calculated using records from 1992–2012. For both estimates, the area of occupancy is likely significantly under-estimated due to limited sampling across the occupied range.
Trend	Contracting			The AOO has declined since European settlement, due to habitat loss induced by land clearing, fragmentation, timber harvesting and fire. This includes the local extinction of some subpopulations (Woinarski et al. 2014). AOO is likely to continue contracting due to loss of suitable habitat resulting from the 2019–20 bushfires, planned burning, land clearing and timber harvesting. See Table 1 for further information.

Metric	Estimate used in the assessment	Minimum plausible value	Maximum plausible value	Justification
Number of subpopulations	Unknown	Unknown	Unknown	The subspecies is wide-ranging and is known from many sites throughout Qld, NSW and Vic. Therefore, the number of subpopulations is not able to be estimated.
Trend	contracting			The number of subpopulations is likely to be declining based on the factors decreasing the AOO and EOO.
Basis of assessment of subpopulation number	There is no information on the number of subpopulations throughout the subspecies range. The number of subpopulations is likely to be large, considering the large AOO and EOO.			
No. locations	Unknown	Unknown	>10	The number of locations is not known with any certainty. The 2019–2020 bushfire events burnt a large area of Eastern Australia (100 000 km^2), overlapping c. 41% of the yellow-bellied glider (south-eastern) distribution (Legge et al 2021). However, the fire severity was highly spatially variable, with yellow-bellied glider (south-eastern) persisting in at least some burnt areas. Thus, the number of locations may be significantly greater than 10.
Trend	contracting			The severity, frequency and scale of catastrophic bushfires will likely increase due to climate change. Therefore, the number of locations in which a single bushfire can rapidly affect all individuals will likely decrease.
Basis of assessment of location number	The subspecies occurs across much of the land in four states and territories. A large number of bushfires are likely to be required to impact all individuals.			
Fragmentation	Not severely fragmented– less than 50% of AOO in habitat patches that cannot support minimum viable population.			
Fluctuations	Not subject to extreme fluctuations in EOO, AOO, number of subpopulations, locations or mature individuals.			

Criterion 1 Population size reduction

Reduction in total numbers (measured over the longer of 10 years or 3 generations) based on any of A1 to A4			
	Critically Endangered **Very severe reduction**	**Endangered** **Severe reduction**	**Vulnerable** **Substantial reduction**
A1	≥ 90%	≥ 70%	≥ 50%
A2, A3, A4	≥ 80%	≥ 50%	≥ 30%

A1 Population reduction observed, estimated, inferred or suspected in the past and the causes of the reduction are clearly reversible AND understood AND ceased.

A2 Population reduction observed, estimated, inferred or suspected in the past where the causes of the reduction may not have ceased OR may not be understood OR may not be reversible.

A3 Population reduction, projected or suspected to be met in the future (up to a maximum of 100 years) [*(a) cannot be used for A3*]

A4 An observed, estimated, inferred, projected or suspected population reduction where the time period must include both the past and the future (up to a max. of 100 years in future), and where the causes of reduction may not have ceased OR may not be understood OR may not be reversible.

Based on any of the following

(a) direct observation [except A3]
(b) an index of abundance appropriate to the taxon
(c) a decline in area of occupancy, extent of occurrence and/or quality of habitat
(d) actual or potential levels of exploitation
(e) the effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites

Criterion 1 evidence

Eligible under Criterion 1 A4(c) for listing as Vulnerable

Generation length

The generation length of the yellow-bellied glider (south-eastern) is estimated to be four to five years (Woinarski et al. 2014), giving a timeframe of 12–15 years for this criterion (three generations).

Past decline due to habitat loss, fragmentation, forestry and altered fire regimes

Prior to the 2019–20 bushfires, trends of population decline, and the rate of this decline, could not be reliably estimated. There has been no integrated long-term monitoring program across major subpopulations, and the inherently low density of the subspecies throughout its broad range can make subpopulations difficult to identify and count. Woinarski et al. (2014) suggested that the population is declining due to habitat loss through fragmentation, clearing, bushfires, and some forestry practices (see Table 1 for more information on threats). The requirement for specific habitat characteristics, including large and socially exclusive home ranges and mature eucalypt forests, has led to a decline and patchy distribution of gliders throughout their range (Goldingay & Possingham 1995). The subspecies has likely declined significantly in the wet ash forests of the Central Highlands of Victoria over the past 25 years, though were too uncommon to facilitate detailed statistical analyses in a recent multi-decadal study (Lindenmayer et al. 2020). Across the subspecies range, there have been localised declines due to clearing and timber harvesting (Milledge et al. 1991; Andrews et al. 1994; Smith et al. 1994; Kavanagh & Webb 1998; Lindenmayer et al. 1999b;), as well as extinctions of some subpopulations due to unknown causes (e.g., Booderee National Park) (Woinarksi et al. 2014). However, it is also noted that long-term studies, particularly in south-eastern NSW, have shown that glider populations fluctuate greatly over time (Kavanagh et al. 2021).

Subpopulation-level studies over the past three generations include a study on a subpopulation on the Bago Plateau (southern NSW) that indicated it was declining at a rate of over 30 percent over a three-generation period. The reason for this decline is unknown, and the suggested impact of timber harvesting was not found to be significant (Kambouris et al. 2013). In contrast, a study surveying a subpopulation in Richmond Range National Park (north-east NSW) found near constant abundance from 2014 to 2016, with no change in yellow-bellied glider (south-eastern) site occupancy (Goldingay et al. 2017). This subpopulation was not affected by the 2019–20 bushfires and appears to have undergone no recent decline despite drought conditions (Goldingay 2020. pers comm 15 December). It is notable that Richmond Range National Park is protected, and thus decline due to timber harvesting and clearing is likely not represented in these trends.

Kavanagh et al. (2021) observed that fire-affected populations of the yellow-bellied glider appeared to have declined at low elevation sites in north-eastern NSW. There was a negative relationship between detectability and the number of fires since 1990, though Kavanagh (2021) suggests declines in populations appear to have occurred independently of fire (Kavanagh 2021). Surveys revealed that only 25 percent of previously utilized sites were occupied, and yellow-bellied gliders (South-eastern) were not observed at some unburnt sites. Furthermore, a number of sap trees that had previously been frequently used appeared to have no evidence of recent yellow-bellied glider (South-eastern) activity (R Kavanagh 2021. pers comm 10 August). The driving factors behind these potential declines are unclear, though climate change may be responsible through reductions to rainfall and increases in mean daytime and night-time temperatures (R Kavanagh 2021. pers comm 10 August), similarly to the related greater glider (Smith & Smith 2018, 2020; Wagner et al. 2020). Further research is required to determine the mechanisms behind these observations of non-fire related population decline, and how they are impacting on the total population (R Kavanagh 2021. pers comm 10 August).

Past decline over three generations may perhaps approach 30 percent, as suggested by Woinarski et al. (2014), and this threshold has likely been eclipsed after the 2019–20 bushfires. Further research on the subspecies' population dynamics over a large segment of the distribution will be useful to accurately determine population decline (Goldingay et al. 2017).

Future decline due to habitat loss, fragmentation, forestry, altered fire regimes and range contraction

The combination of fragmentation, bushfires, drought, forestry, and range contraction is likely to lead to decline in the yellow-bellied glider (south-eastern) over the next three generations (see Table 1). It is notable that the impacts of timber harvesting may lessen across the Vic distribution of the yellow-bellied glider (south-eastern) due to the scheduled cessation of native forest timber harvesting in 2030 and use of new harvesting techniques (VicForests 2021. pers comm 24 June). However, ongoing legacy impacts of fragmentation and past clearfell timber harvesting with little regeneration will likely still lead to some population decline for the subspecies.

Impacts of the 2019–20 bushfires

In 2019–20, following years of drought (DPI 2020), catastrophic bushfire conditions resulted in extensive bushfires covering an unusually large area of eastern Australia. Recent analysis suggests that 41 percent of the subspecies distribution was affected by the fires (Legge et al. 2021). 54 percent of species records in NSW were in the fire affected areas (DPIE 2020). It is suspected that the total population will continue to decline after the fires due to post-fire effects which include the loss of important habitat features (Lunney 1987; Goldingay & Kavanagh 1991) and post-fire salvage timber harvesting (Noss & Lindenmayer 2006; Lindenmayer et al. 2008).

<u>On-ground surveys</u>

On-ground surveys show that the fires had a substantial impact on the yellow-bellied glider (south-eastern). In surveys of 30 sites in East Gippsland, the subspecies was present in highest abundance at unburned sites and sites with low canopy scorching and was absent at sites with high or complete canopy scorching (Burns 2020. pers comm 15 December). Surveys in the Shoalhaven Area (NSW) suggest that subpopulations in canopy-impacted sites underwent severe decline or extinctions (Craven & Daly 2020). In all surveyed areas, yellow-bellied gliders (south-eastern) were not detected at high or extreme fire severity sites, and 68 percent of transects had fewer individuals detected than before the bushfires (Craven & Daly 2020). Surveys of the lower Richmond and Clarence floodplain by the Nature Conservation Council of NSW recorded yellow-bellied glider (south-eastern) vocalisations at six of the 30 monitored sites prior to the spring fires of 2019, though no vocalisations have been recorded since the 2019–20 bushfires. It is estimated that 20 percent of large hollow bearing trees at these sites have been lost (NCC 2021).

Surveys in 2021 investigated the impacts of the 2019–20 bushfires on yellow-bellied glider (south-eastern) subpopulations in north-eastern and south-eastern NSW, both regions having previously been surveyed for the subspecies (Kavanagh et al. 2021). The north-eastern NSW study sites have a history of both timber harvesting and fire (Kavanagh et al. 1995; Kavanagh & Stanton 2005; McLean et al. 2015; McLean et al. 2018). Surveys of these sites in 2020–2021 observed yellow-bellied gliders (south-eastern) at 10 of the 47 sites where they had been recorded previously (21.3 percent), and at one additional site where they had not been recorded previously. The relative abundance of the subspecies followed a slight negative trend in relation to the proportion of severely burnt forest across the 47 sites where they had been previously recorded, and there was also a slight positive relationship between abundance and the proportion of unburnt and low severity burnt forest within the local landscape around each site (Kavanagh et al. 2021).

The south-eastern NSW sites have a history of timber harvesting but historically low fire frequency (Kavanagh et al. 2021). The northernmost eight of the 18 sites surveyed had been surveyed multiple times previously, with yellow-bellied gliders (south-eastern) recorded at just two sites prior to 2021. One of these sites has only one record of the subspecies, with no observations in subsequent surveys. Surveys in 2021 observed yellow-bellied gliders (south-eastern) at one of the two sites where they were previously found, as well as two new sites (including a severely burnt site). The impact of fires on this subpopulation is difficult to determine, though the subspecies is still present and able to be observed. The other subset of ten south-east study lines (the Waratah Creek grid) had also been surveyed previously on multiple occasions (Kavanagh 1984; Kavanagh & Webb 1998). 77 percent of the grid was burnt at moderate (42 percent), high (30 percent) or extreme severity (5 percent), with the remaining 23 percent either unburnt (3 percent) or burnt at low severity (20 percent). Previously, the 'detectable subpopulation' (an index) of yellow-bellied gliders (south-eastern) (total number ~ 625) fluctuated between 21–63 individuals throughout all years of the 40-year study. However, surveys in 2021 found only three individuals, representing a decline in observations of 86–97 percent. It is highly likely that this observed decline in the yellow-bellied glider (south-eastern) subpopulation at Waratah Creek is due to the impacts of fire (Kavanagh et al. 2021).

Table 4: Summary of on-ground survey results for yellow-bellied glider (south-eastern)

Locality	Survey date	Number of sites	Impact of mild fire	Impact of Severe fire	Source
East Gippsland	September – October 2020	30 sites (19 with pre-fire detections)	Reduced number of sightings	Absent (100% decline)	P. Burns; unpublished
Shoalhaven	May – June 2020	71 sites (31 with pre-fire detections)	Unclear. Detected at 10/31 sites where it was previously found, all burnt at low-moderate severity. Unclear how many sites previously occupied by the subspecies were burnt at each severity. Likely that occupancy was reduced at mildly burnt sites.	Absent (100% decline)	Craven P & Daly G (2020)
Lower Richmond and Clarence floodplains	Unknown	30 sites (6 with pre-fire detections)	Unknown	Absent (100% decline)	NCC (pers comm. 24 June 2021)
North-east NSW (at sites with a history of forestry and high fire frequency)	November 2020, April - May 2021	94 sites (47 with pre-fire detections)	Found at only 21.3% of sites they were previously present at (and one additional site). Less likely to be present at sites in severely burned forest		Kavanagh et al. (2021)
South-east NSW (sites with a history of forestry and low fire frequency)	May 2021	8 sites (2 with pre-fire detections)	Recorded at two new sites and one site with previous records. Not recorded at different a site with previous records.		
South-east NSW (sites with a history of forestry and low fire frequency)	May 2021	1 large 100 ha site (10 transects, all with pre-fire detections)	3 individuals recorded after the 2019–20 fires, totalling 86-97% decline across all transects. 77% of site burn at moderate (42%), high (30%), or extreme (5%) severity. 20% burnt at low severity, 3% unburnt.		

Across the south-eastern and north-eastern sites, there was an observed negative relationship between yellow-bellied glider (south-eastern) abundance and increasing fire severity in the local landscape (Kavanagh et al. 2021). It is considered likely that observations of the subspecies in severely burnt areas were due to the proximity of unburnt or low severity burnt areas nearby. However, univariate analyses were unable to fully explain the importance of patchiness in fire severity in the local landscape (Kavanagh et al. 2021).

<u>Expert elicitation</u>

In a project run by the Threatened Species Recovery Hub (Legge et al. 2021), expert elicitation was used to estimate the extent of population decline after fires of varying severity, and the predicted population trajectories out to three generations after the 2019–20 fires. Information on population response to fires of varying severity was combined with spatial estimates of the overlaps between the subspecies distribution and fire severity mapping. The analysis suggests that site-level decline at severely affected sites is around 81.6 percent (80 percent confidence bounds: 67.7 to 90.4 percent) one week after the fires and 78.1 percent (80 percent confidence bounds: 91 and 62.6 percent) three generations after the fires. These estimates are consistent with the site-level empirical data summarised above (Table 4). Considering the proportion of the distribution burnt in low and high severity fire, *the overall* population of yellow-bellied gliders (south-eastern) was estimated to have declined by 21 percent one year after the fire but may have declined by as much as 29 percent (the lower 80 percent confidence bound). By comparing this trajectory to that predicted for subpopulations that were not exposed to fires, the elicitation indicated that after one-year, yellow-bellied glider (south-eastern) populations would be around 19.5 percent lower than they would have been, had the fires not occurred. In other words, the 2019-20 fires will have caused an additional 19.5 percent decline on top of any pre-existing declines.

By three generations after the 2019–20 bushfires, the overall population is predicted to be 25 percent lower than its pre-2019 level, but possibly as much as 38 percent lower (80 percent confidence bound). By comparing this trajectory to that predicted for subpopulations that were not exposed to fires, the elicitation indicated that after three generations, the fires caused an additional 15 percent decline on top of 10 percent overall decline due to pre-existing factors. These elicitations assumed no further extensive fire events in the range of the yellow-bellied glider (south-eastern) over the 3-generation period.

Overall population decline

The yellow-bellied glider (south-eastern) is declining in abundance due to the catastrophic 2019–20 bushfires, and ongoing habitat loss from clearing, fragmentation, bushfires, drought, and some forestry practices. It may also be declining due to threats associated with climate change (R Kavanagh 2021. pers comm 10 August). Overall decline over a period including both the past and the future (2019–2031/2034) is estimated by Legge et al. (2021) at 25 percent, or up to 38 percent. However, given that large-scale fire and catastrophic drought were not accounted for during projection of future declines , and such events are predicted to increase in frequency (CSIRO & Bureau of Meteorology 2015), it is likely that this decline exceeds 25 percent and is closer to the lower bounds of 38 percent (Legge et al. 2021).

Conclusion

The Committee considers that the subspecies has undergone a substantial reduction in numbers over three generations (12–15 years for this assessment) which is equivalent to at least 30 percent. The reduction has not ceased, and the cause has not ceased. Therefore, the subspecies has met the relevant elements of Criterion 1 to make it eligible for listing as Vulnerable.

Criterion 2 Geographic distribution as indicators for either extent of occurrence AND/OR area of occupancy

		Critically Endangered Very restricted	Endangered Restricted	Vulnerable Limited
B1.	Extent of occurrence (EOO)	**< 100 km²**	**< 5,000 km²**	**< 20,000 km²**
B2.	Area of occupancy (AOO)	**< 10 km²**	**< 500 km²**	**< 2,000 km²**
AND at least 2 of the following 3 conditions:				
(a)	Severely fragmented OR Number of locations	**= 1**	**≤ 5**	**≤ 10**
(b)	Continuing decline observed, estimated, inferred or projected in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) area, extent and/or quality of habitat; (iv) number of locations or populations; (v) number of mature individuals			
(c)	Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or populations; (iv) number of mature individuals			

Criterion 2 evidence

Not eligible

The extent of occurrence (EOO) is estimated at 1,285,082 km² and the area of occupancy (AOO) is estimated at 12,724 km². These figures are based on the mapping of point records from a 20-year period (2000–2020), obtained from state governments, museums, and CSIRO. The EOO was calculated using a minimum convex hull, and the AOO calculated using a 2x2 km grid cell method, based on the IUCN Red List Guidelines 2014 (IUCN 2019). The AOO is likely significantly under-estimated due to limited sampling across the occupied range (Woinarski et al. 2014).

Conclusion

Following assessment of the data the Committee considers that the subspecies is not eligible for listing in any category under this criterion as neither the EOO or AOO are likely to be limited.

Criterion 3 Population size and decline

		Critically Endangered Very low	Endangered Low	Vulnerable Limited
Estimated number of mature individuals		< 250	< 2,500	< 10,000
AND either (C1) or (C2) is true				
C1. An observed, estimated or projected continuing decline of at least (up to a max. of 100 years in future)		**Very high rate** **25% in 3 years or 1 generation** **(whichever is longer)**	**High rate** **20% in 5 years or 2 generation** **(whichever is longer)**	**Substantial rate** **10% in 10 years or 3 generations** **(whichever is longer)**
C2. An observed, estimated, projected or inferred continuing decline AND its geographic distribution is precarious for its survival based on at least 1 of the following 3 conditions:				
(a)	(i) Number of mature individuals in each population	≤ 50	≤ 250	≤ 1,000
	(ii) % of mature individuals in 1 population =	90 – 100%	95 – 100%	100%
(b) Extreme fluctuations in the number of mature individuals				

Criterion 3 evidence

Not eligible

There is no reliable estimate of population size, though Woinarski et al. (2014) estimated that the number of mature individuals was greater than 100,000. However, due to ongoing decline and the 2019–20 bushfires, the current population size is likely below 100,000 mature individuals, but still substantially greater than 10,000 mature individuals.

Conclusion

Following assessment of the data the Committee considers that the species/subspecies is not eligible for listing in any category under this criterion as the total population size is not likely to be limited.

Criterion 4 Number of mature individuals

	Critically Endangered **Extremely low**	**Endangered** **Very Low**	**Vulnerable** **Low**
D. Number of mature individuals	< 50	< 250	< 1,000
D2.[1] *Only applies to the Vulnerable category* Restricted area of occupancy or number of locations with a plausible future threat that could drive the species to critically endangered or Extinct in a very short time			D2. Typically: area of occupancy < 20 km^2 or number of locations ≤ 5

[1] The IUCN Red List Criterion D allows for species to be listed as Vulnerable under Criterion D2. The corresponding Criterion 4 in the EPBC Regulations does not currently include the provision for listing a species under D2. As such, a species cannot currently be listed under the EPBC Act under Criterion D2 only. However, assessments may include information relevant to D2. This information will not be considered by the Committee in making its recommendation of the species' eligibility for listing under the EPBC Act, but may assist other jurisdictions to adopt the assessment outcome under the *common assessment method*.

Criterion 4 evidence

Not eligible

As described above under Criterion 3, there is no reliable estimate of the yellow-bellied glider (south-eastern) population size. Due to ongoing decline and the 2019–20 bushfires, the current population size is likely below 100,000 mature individuals but still substantially above 10,000 mature individuals. Therefore, the subspecies has not met this required element of this criterion. The species has an AOO of above 20 km^2 and more than five locations, so is not eligible under criterion D2.

Conclusion

Following assessment of the data the Committee considers that the species/subspecies is not eligible for listing in any category under this criterion as total population size is not likely to be limited.

Criterion 5 Quantitative analysis

	Critically Endangered **Immediate future**	**Endangered** **Near future**	**Vulnerable** **Medium-term future**
Indicating the probability of extinction in the wild to be:	≥ 50% in 10 years or 3 generations, whichever is longer (100 years max.)	≥ 20% in 20 years or 5 generations, whichever is longer (100 years max.)	≥ 10% in 100 years

Criterion 5 evidence

Insufficient data to determine eligibility

A population viability analysis has been completed to assess the minimum viable number of populations and habitat area (Goldingay & Possingham 1995), but an assessment of the likelihood of extinction has not been assessed.

Conclusion

Population viability analysis has not been undertaken. Therefore, there is insufficient information to determine the eligibility of the subspecies for listing in any category under this criterion.

Adequacy of survey

The survey effort has been considered adequate and there is sufficient scientific evidence to support the assessment.

Public consultation

Notice of the proposed amendment and a consultation document was made available for public comment for 36 business days between 6 May 2021 and 24 June 2021.

Listing and Recovery Plan Recommendations

The Threatened Species Scientific Committee recommends:

(i) that the list referred to in section 178 of the EPBC Act be amended by **including** *Petaurus australis australis* in the list in the Vulnerable category.

(ii) that there should be a recovery plan for this subspecies.

Appendix

Appendix A: Identified tree species found in the habitat of the yellow-bellied glider (south-eastern)

Scientific Name	**Common Name/s**	**Used as a Sap tree**
Acacia mabellae	Mabel's wattle	Yes
A. mearnsii	black wattle	Yes
Angophora subvelutina	broad-leaved apple	Yes
A. leiocarpa	rusty gum	Yes
Corymbia citriodora	spotted gum, lemon-scented gum	Yes
C. gummifera	red bloodwood	Yes
C. henryi	large-leaved spotted gum	Yes
C. intermedia	pink bloodwood	Yes
C. maculata	spotted gum	Yes
C. trachyphloia	brown bloodwood	Unknown
Eucalyptus acmenoides	white mahogany	Unknown
E. amplifolia	cabbage gum	Yes
E. andrewsii	New England blackbutt, gum-topped peppermint	Yes
E. angophoroides	apple-topped box	Yes
E. bancroftii	orange gum, Bancroft's red gum	Yes
•*E baxteri*	brown stringybark	Unknown
E. biturbinata	grey gum	Yes
E. bosistoana	coast grey box	Yes
E. botryoides	Bangalay, southern mahogany	Yes
E. campanulata	New England blackbutt	Unknown
E. crebia	narrow-leaved ironbark	Unknown
E. cypellocarpa	monkey gum, mountain grey Gum	Yes
E. dalrympleana	mountain gum	Yes
E. deanei	mountain blue gum, round-leaved gum	Yes
E. dunnii	white gum	Yes
E. elata	red peppermint	Unknown
E. eugenioides (includes *E. nigra*)	thin-leaved stringybark	Yes
E. fastigata	brown Barrel, cut-tail	Yes
E. fibrosa	broad-leaved ironbark	Unknown
E. fraxinoides	white ash	Unknown
E. grandis	flooded gum, rose gum	Yes
E. laevopinea	silvertop stringybark	Yes
E. longirostrata	grey gum	Yes
E. major	grey gum	Yes
E. melliodora	yellow box	Yes

E. microcorys	tallowwood	Unknown
E. moluccana	grey box	Yes
E. nobilis	ribbon gum	Unknown
E. obliqua	messmate	Yes
E. ovata	swamp gum	Yes
E. pauciflora	snow gum	Unknown
E. pilularis	blackbutt	Yes
E. pilularis/C. maculata hybrid		Yes
E. piperita	Sydney peppermint	Yes
E. propinqua	grey gum	Yes
E. punctata	grey gum	Yes
E. racemosa	narrow-leaved scribbly Gum	Yes
E. radiata	narrow-leaved peppermint	Unknown
E. regnans	mountain ash	Unknown
E. resinifera	red mahogany	Yes
E. robusta	swamp mahogany	Unknown
E. saligna	Sydney blue gum	Yes
E. sclerophylla	hard-leaved scribbly gum	Yes
E. seeana	narrow-leaved red gum	Yes
E. sieberi	silvertop ash	
E. signata	scribbly gum	Yes
E. siderophloia	grey ironbark	Unknown
E. sphaerocarpa	Blackdown stringybark	Yes
E. tereticornis	forest red gum	Yes
E. viminalis	ribbon gum, manna gum	Yes
Lophostemon confertus	brush box	Yes
Syncarpia glomulifera	turpentine	

Information sourced from (Kavanagh 1987a; Eyre & Smith 1997; NSW NPWS 2002; Eyre 2004; Eyre 2005; Eyre & Goldingay 2005; Irish & Kavanagh 2011).

Cataloguing data

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