Appendix L Species Conservation Advice

Approved Conservation Advice for *Rostratula australis* (Australian painted snipe)

(s266B of the *Environment Protection and Biodiversity Conservation Act 1999*)

This Conservation Advice has been developed based on the best available information at the time this Conservation Advice was approved; this includes existing plans, records or management prescriptions for this species.

Description

Rostratula australis (Australian painted snipe), Family Rostratulidae, is a stocky wading bird approximately 240–300 mm in length, with a wingspan of 500–540 mm and weighing 125–130 g (Birds Australia, 2012). The adult female is more colourful and larger than the male. It has a chocolate-brown head with chestnut patch in the nape, a comma-shaped white marking around the eye and metallic green back and wings, densely barred olive and black (Rogers pers. comm., 2012). A diagnostic white 'harness marking' runs from the mantle onto the breast (Rogers pers. comm., 2012). It has a brown eye, white belly, bluish-green legs and long pink-orange bill darkening towards the tip (Reader's Digest, 1997). The male is smaller than the female and has a duller head pattern (Rogers pers. comm., 2012). It has a mottled grey-brown head and neck, with buff stripe down the centre of the crown and through the eyes. Wings and back are barred black, buff and white, and the breast has a broad black band (Reader's Digest, 1997). There is no seasonal variation in the plumage of the Australian painted snipe. The juvenile is separable though very similar to the adult male (Marchant and Higgins, 2003).

Conservation Status

The Australian painted snipe is listed as **endangered** under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act). This species is eligible for listing as endangered as it is inferred to have undergone a severe decline in the number of mature individuals in excess of 50% over the last three generations (~26 years) associated with wetland loss and degradation (TSSC, 2012).

The Australian painted snipe is also listed as a marine species (as *Rostratula benghalensis*) and a migratory species (under the China-Australia Migratory Bird Agreement as *Rostratula benghalensis*) under the EPBC Act.

The species is listed as threatened under various state and territory lists and legislation:

State	List/legislation	Listing status	Listed name
Queensland	*Nature Conservation (Wildlife) Regulations 2006*	vulnerable	*Rostratula australis*
New South Wales	*Threatened Species Conservation Act 1995*	endangered	*Rostratula benghalensis australis*
Victoria	*Flora and Fauna Guarantee Act 1988 – Threatened List – October 2010*	threatened	*Rostratula australis*
South Australia	*National Parks and Wildlife Act 1972*	vulnerable	*Rostratula benghalensis*
Western Australia	*Wildlife Conservation (Specially Protected Fauna) Notice 2010(2)*	rare or likely to become extinct	*Rostratula benghalensis australis*
	Threatened and Priority Fauna ranking	vulnerable	
Northern Territory	*Territory Parks and Wildlife Conservation Act 2000*	vulnerable	*Rostratula benghalensis australis*

Cultural Significance

The Australian painted snipe is not known to be culturally significant.

Distribution and Habitat

The Australian painted snipe occurs in shallow freshwater (occasionally brackish) wetlands, both ephemeral and permanent, such as lakes, swamps, claypans, inundated or waterlogged grassland/saltmarsh, dams, rice crops, sewage farms and bore drains, generally with a good cover of grasses, rushes and reeds, low scrub, *Muehlenbeckia* spp. (lignum), open timber or samphire (Reader's Digest, 1997; Marchant and Higgins, 2003). It has been recorded at wetlands in all states and territories (Barrett et al, 2003; Blakers et al., 1984) and is most common in eastern Australia.

Important areas for this species in the past have included the Murray-Darling Basin (particularly the Riverina of Victoria and New South Wales), Queensland Channel Country, Fitzroy Basin of Central Queensland, south-eastern South Australia and adjacent parts of Victoria (Rogers et al., 2005). Records published over the past twenty years provide evidence for Australian painted snipe occurring more widely and frequently in the remote arid and tropical regions of Australia than was previously thought (Hassell and Rogers, 2002; Jaensch 2003a, 2003b; Jaensch et al., 2004; Black et al., 2010).

The Australian painted snipe is inferred to have undergone a severe decline in the number of mature individuals since the 1950s (Garnett and Crowley, 2000; Lane and Rogers, 2000; Rogers et al., 2005; Garnett et al., 2011; BirdLife Australia, 2012) and specifically over the last three generations (~26 years) due to the loss and degradation of its wetland habitat (Rogers et al., 2005). There has been an increase in the number of sightings in 2010–11 associated with increased rainfall; however, this must be considered within the context of overall, long-term population decline (Jaensch pers. comm., 2012; BirdLife Australia, pers. comm., 2012; Rogers pers. comm., 2012). It is estimated that the species' current population is 2500 mature individuals (Garnett et al., 2011; BirdLife Australia, pers. comm., 2012).

The species is widespread and is not considered to have a limited geographic distribution. Its current extent of occurrence estimated to be 7,100,000 km^2 and stable (Garnett et al., 2011). The species' area of occupancy was estimated by Garnett et al. (2011) to be 2000 km^2 and decreasing; however, given the exceptional rainfall of 2010-11 this figure is currently assumed to be higher. The Australian painted snipe occurs within many Natural Resource Management (NRM) Regions and Interim Biogeographic Regionalisation for Australia (IBRA) Bioregions across Australia.

The distribution of this species overlaps with a number of EPBC Act-listed threatened ecological communities, including Seasonal Herbaceous Wetlands (Freshwater) of the Temperate Lowland Plains and Upland Wetlands of the New England Tablelands and the Monaro Plateau.

The Department of Sustainability, Environment, Water, Population and Communities has prepared survey guidelines for Australia's threatened birds (Commonwealth of Australia, 2010). These survey guidelines are intended to provide guidance for stakeholders on the effort and methods considered appropriate when conducting a presence/absence survey for listed threatened species.

Threats

The main identified threat to the Australian painted snipe is the loss and degradation of wetlands, through drainage and the diversion of water for agriculture and reservoirs (Lane and Rogers 2000; Garnett et al., 2011). Rogers et al. (2005) state that the loss of breeding habitat in the Murray-Darling Basin has occurred through: (1) the reduced frequency of flooding in previously suitable habitat, exacerbated by a loss of fresh water to irrigation and other diversions; (2) water levels being stabilised in remaining wetlands so that water becomes too deep, or continuous reed beds develop; and (3) changes to vegetation through increased cropping, and possibly through altered fire regimes at some sites. These hydrological changes have occurred in parallel with an extended period of drought in Australia (BoM, 2010) and these conditions have intensified the impacts of wetland degradation and water diversion in the Murray-Darling Basin.

Grazing and the associated trampling of wetland vegetation/nests, nutrient enrichment and disturbance to substrate by livestock may threaten the Australian painted snipe in certain regions, particularly where grazing is concentrated around wetlands during dry seasons (Johnstone and Storr, 1998; Rogers et al., 2005; Jaensch pers. comm., 2012).

Reduced rainfall and runoff in the Murray-Darling Basin associated with climate change (CSIRO 2008, 2011) may threaten the Australian painted snipe in the future. The species is strongly affected by seasonal conditions and appears to depend on the Murray-Darling Basin for breeding; as such, these conditions could have a significant impact on the species if combined with other known and potential threats.

Predation by feral animals (e.g. nest predation by foxes (*Vulpes vulpes*) or cats (*Felis catus*)) may be a threat to the Australian painted snipe, however there is no evidence for this. Additional potential threats include coastal port and infrastructure development, shale oil mining near autumn-winter sites for this species on the central Queensland coast (Houston and Black, pers. comm., 2012) and the replacement of native wetland vegetation by invasive weeds (Rogers et al., 2005). The impacts of fire on the Australian painted snipe are unknown, but may have either a positive or negative influence (Rogers et al., 2005).

Research Priorities

Research priorities that would inform future regional and local priority actions include:

- Support and enhance existing programs for the Australian painted snipe that are managed by BirdLife Australia.
- Continue to monitor the species to more precisely assess population size, distribution and the relative impacts of threatening processes.
- Identify and describe the ecological and hydrological character of sites that are suitable for the Australian painted snipe, particularly those known to be used by the species for breeding.
- Investigate potential food resources for the species and monitor changes to the abundance and diversity of these resources (e.g. invertebrates).
- Directly monitor the breeding and non-breeding behaviour of the Australian painted snipe with the use of radio transmitters and/or tagging methods.

Regional Priority Actions

The following regional priority recovery and threat abatement actions can be done to support the recovery of the Australian painted snipe.

<u>Habitat Loss, Disturbance and Modification</u>

- Develop management guidelines for breeding and non-breeding habitat.
- Monitor the progress of recovery, including the effectiveness of management actions and the need to adapt them if necessary.
- Ensure there is no disturbance in areas where the species is known to breed, excluding necessary actions to manage the conservation of the species.
- Control access routes to suitably constrain public access to existing and future breeding sites on public land.
- Suitably control and manage access on private land and other land tenure.

- Minimise adverse impacts from land use at known sites.
- Manage any changes to hydrology that may result in changes to water table levels, run-off, salinity, algal blooms, sedimentation or pollution.
- Manage any disruptions to water flows.
- Investigate formal conservation arrangements, management agreements and covenants on private land, and for crown and private land investigate/secure inclusion in reserve tenure if possible.
- Manage any other known, potential or emerging threats including inappropriate fire regimes and coastal port/infrastructure development.

<u>Invasive Weeds</u>

- Implement the Parkinsonia (*Parkinsonia aculeata*) Strategic Plan (Commonwealth of Australia, 2000) for the control of this species within the range of the Australian painted snipe.
- Identify and remove weeds in wetland areas that could become a threat to the Australian painted snipe, using appropriate methods.
- Ensure chemicals or other mechanisms used to eradicate weeds do not have a significant adverse impact on the Australian painted snipe.

<u>Trampling, Browsing or Grazing</u>

- Develop and implement a stock management plan for roadside verges and travelling stock routes which include swamps, marshes or wetlands.
- If livestock grazing occurs in known Australian painted snips habitats, ensure land owners/managers use an appropriate management regime and density that does not detrimentally affect Australian painted snipe nesting.
- If appropriate, manage total grazing pressure at important breeding sites through exclusion fencing or other barriers.

<u>Animal Predation or Competition</u>

- Implement the national threat abatement plans for the European red fox (DEWHA, 2008a) and feral cats (DEWHA, 2008b) to control the adverse impacts of foxes (*Vulpes vulpes*) and cats (*Felis catus*) in the species' range.
- Continue baiting to control population numbers of feral animals.

<u>Fire</u>

- Develop and implement a suitable fire management strategy for the habitat of the Australian painted snipe.

<u>Conservation Information</u>

- Raise awareness of the Australian painted snipe within the local community and the importance of reporting observations to BirdLife Australia, using fact sheets and/or brochures.
- Advertise and encourage use of Australian painted snipe survey techniques and survey forms (Birds Australia, 2012).
- Organise field days with industry and interest groups to raise awareness and share information on the species. These groups may include natural resource management groups, catchment management authorities, Indigenous groups, conservation organisations, local and state governments, and private landholders.
- Engage with private landholders and land managers responsible for the land on which populations occur and encourage these key stakeholders to contribute to the implementation of conservation management actions.
- Raise awareness of banded individuals (see BirdLife Australia, 2012) to increase the likelihood of re-sighting and reporting.
- Facilitate the exchange of information between interested parties, including sightings, research and management approaches.

This list does not necessarily encompass all actions that may be of benefit to the Australian painted snipe, but highlights those that are considered to be of highest priority at the time of preparing the Approved Conservation Advice.

Existing Plans/Management Prescriptions that are Relevant to the Species

- Australian Painted Snipe Project (BirdLife Australia, 2012).
- Draft National Recovery Plan for the Australian Painted Snipe *Rostratula australis* 2005-2010 (Compiled by the Victorian Department of Sustainability and Environment for the Australian Government Department of the Environment and Heritage, June 2005).
- Threat abatement plan for predation by the European red fox (Commonwealth of Australia, 2008a).
- Threat abatement plan for predation by feral cats (Commonwealth of Australia, 2008b).
- Australian painted snipe survey form, survey instructions, brochure and newsletters (Birds Australia, 2012).

These prescriptions were current at the time of publishing; please refer to the relevant agency's website for any updated versions.

References cited in the advice

Barrett G, Silcocks A, Barry S, Cunningham R and Poulter R (2003). The new atlas of Australian birds. Royal Australasian Ornithologists Union, Hawthorn East, Victoria.

BirdLife Australia (2012). Painted Snippets: the newsletter of the Australian painted snipe project. Volume 4, May 2012.
Viewed: 11 May 2012
Available on the Internet at:
http://www.birdlife.org.au/documents/APS-snippetsMay12.pdf

BirdLife Australia (2012). Personal communication by email, 7 February 2012. BirdLife Australia acknowledges the input and information provided by the following individuals: C. Tzaros, A. Silcocks, D. Ingwersen, J. Thomas, C. Purnell, D. Rogers, A. Geering and D. Parker.

Birds Australia (2012). Australian painted snipe *Rostratula australis*.
Viewed: 11 May 2012
Available on the Internet at:
http://www.birdsaustralia.com.au/our-projects/australian-painted-snipe.html

Black R, Houston W and Jaensch R (2010). Evidence of regular seasonal migration by Australian painted snipe *Rostratula australis* to the Queensland tropics in autumn and winter. Stilt 58:1–9.

Blakers M, Davies S J J F and Reilly PN (1984). The atlas of Australian birds. Royal Australasian Ornithologists Union. Melbourne University Press, Victoria.

Bureau of Meteorology (BoM) (2010). Special climate statement 22: Australia's wettest September on record but it is not enough to clear long-term rainfall deficits. Issued 1 October 2010; updated 5 October 2010. National Climate Centre, Bureau of Meteorology.
Viewed: 11 May 2012
Available on the Internet at:
http://www.bom.gov.au/climate/current/statements/scs22.pdf

Commonwealth of Australia (2000). Weeds of National Significance: Parkinsonia (*Parkinsonia aculeata*) Strategic Plan. Commonwealth of Australia and the National Weeds Strategy Executive Committee, 2000.
Viewed: 27 July 2012
Available on the Internet at: http://www.weeds.org.au/docs/parstrat.pdf

Commonwealth of Australia (2008a). Threat Abatement Plan for predation by the European red fox. Prepared by the Department of Environment, Water, Heritage and the Arts in 2008.
Viewed: 27 July 2012
Available on the Internet at:
http://www.environment.gov.au/biodiversity/threatened/publications/tap/foxes08.html

Commonwealth of Australia (2008b). Threat Abatement Plan for predation by feral cats. Prepared by the Department of Environment, Water, Heritage and the Arts in 2008.
Viewed: 17 July 2012
Available on the Internet at:
http://www.environment.gov.au/biodiversity/threatened/publications/tap/pubs/tap-cat-report.pdf

Commonwealth of Australia (2010). Survey Guidelines for Australia's Threatened Birds. Guidelines for detecting birds listed as threatened under the *Environment Protection and Biodiversity Conservation Act 1999*. Prepared by the Department of Environment, Water, Heritage and the Arts in 2010.
Viewed: 27 July 2012
Available on the Internet at:
http://www.environment.gov.au/epbc/publications/threatened-birds.html

Commonwealth Scientific and Industrial Research Organisation (2008). Rainfall-runoff modelling across the Murray-Darling Basin. A report to the Australian Government from the CSIRO Murray-Darling Basin Sustainable Yields Project, June 2008.
Viewed: 11 May 2012
Available on the Internet at:
http://www.clw.csiro.au/publications/waterforahealthycountry/mdbsy/technical/C-Rainfall-runoffModelling.pdf

Commonwealth Scientific and Industrial Research Organisation (2011). Planning for climate change in the Murray-Darling Basin. 16 March 2006; updated 14 October 2011.
Viewed: 11 May 2012
Available on the Internet at:
http://www.csiro.au/Organisation-Structure/Flagships/Water-for-a-Healthy-Country-Flagship/Murray-Darling-climate-change.aspx

Garnett ST and Crowley GM (2000). The action plan for Australian birds 2000. Environment Australia, Canberra.

Garnett ST, Szabo J and Dutson G (2011). The action plan for Australian birds 2010. CSIRO Publishing, Victoria.

Hassell CJ and Rogers DI (2002). Painted snipe nesting at Taylor's Lagoon, near Broome, north-western Australia. The Stilt 41: 14–21.

Houston W and Black R (2012). Personal communication by email, 7 February 2012. Centre for Environmental Management, Central Queensland University.

Jaensch R (2003a). Breeding by Australian painted snipe in the Diamantina Channel Country, south-western Queensland. The Stilt 43: 20–22.

Jaensch R (2003b). Recent records and breeding of painted snipe *Rostratula benghalensis* in the Mitchell Grass Downs and Sturt Plateau, Northern Territory. NT Naturalist 17: 31–37.

Jaensch R (2012). Personal communication by email, 23 January 2012. Professional Associate, Wetlands international – Oceania.

Jaensch R, McCabe J, Wahl J and Houston W (2004). Breeding by Australian painted snipe on the Torilla Plain, Brigalow Belt coast, Queensland. The Stilt 45: 40–43.

Johnstone RE and Storr GM (1998). Handbook of Western Australia, Volume 1. Non-passerines (Emu to Dollarbird). Western Australian Museum, Perth.

Lane BA and Rogers D (2000). The Australian painted snipe *Rostratula (benghalensis) australis:* An endangered species? Stilt 36: 26–34.

Marchant S and Higgins PJ (eds) (1993). Handbook of Australian, New Zealand and Antarctic Birds. Volume 2: raptors and lapwings. Oxford University Press Australia.

Readers Digest (1997). Complete Book of Australian Birds (second edition). Reader's Digest Australia Pty Ltd, Sydney.

Rogers D (2012). Personal communication by email, 28 February 2012. Arthur Rylah Institute for Environmental Research.

Rogers D, Hnace I, Paton S, Tzaros C, Griffioen P, Herring M, Jaensch R, Oring L, Silcocks A and Weston M (2005). The breeding bottleneck: Breeding habitat and population decline in the Australian painted snipe. Status and Conservation of Shorebirds in the East Asian-Australasian Flyway. Proceedings of the Australasian Shorebirds Conference, 13–15 December 2003, Canberra, Australia.

Threatened Species Scientific Committee (TSSC) (2012). Listing advice for *Rostratula australis* (Australian painted snipe).

The Minister approved this conservation advice and transferred this species from the Critically Endangered to Vulnerable category, effective from 07/12/2016

Conservation Advice

Saccolaimus saccolaimus nudicluniatus

bare-rumped sheathtail bat

Note: The information contained in this conservation advice was primarily sourced from 'The Action Plan for Australian Mammals 2012' (Woinarski et al., 2014). Any substantive additions obtained during the consultation on the draft have been cited within the advice. Readers may note that conservation advices resulting from the Action Plan for Australian Mammals show minor differences in formatting relative to other conservation advices. *These reflect the desire to efficiently prepare a large number of advices by adopting the presentation approach of the Action Plan for Australian Mammals, and do not reflect any difference in the evidence used to develop the recommendation.*

Taxonomy

Conventionally accepted as *Saccolaimus saccolaimus nudicluniatus* (De Vis 1905).

Saccolaimus saccolaimus (Temminck 1838) was first described from Java. It comprises five valid subspecies (Simmons 2005) and is distributed widely from the Solomon Islands and tropical Australia to India (Csorba et al., 2008).

The taxonomic status of the two Australian populations of *Saccolaimus saccolaimus* is unresolved. The taxon *S. s. nudicluniatus* was first described from Queensland (as *Taphozous nudicluniatus*, De Vis 1905). Both the Queensland and Northern Territory (including Kimberley) populations are considered as *S. s. nudicluniatus* under the EPBC Act 1999, but Hall et al. (2008) attributed the Northern Territory population to the nominate *S. s. saccolaimus* of Indonesia. Other previous authors have not attributed the Northern Territory population to either subspecies (McKean et al., 1981; Thomson 1991; Duncan et al., 1999; Schulz & Thomson 2007). Including populations outside Australia, the taxon, *nudicluniatus*, has been considered at the species level (De Vis 1905; Troughton 1925; Corbet & Hill 1980; Nowak & Paradiso 1983), the subspecies level (Koopman 1984, 1994; Flannery 1995, Hall et al., 2008), as well as being synonymised with the nominate (e.g. Goodwin 1979). Its extralimital distribution is also unclear. Flannery (1990) attributed those in New Guinea and the Solomon Islands to *nudicluniatus*, but he later (Flannery 1995) considered that this taxon occurred only in Australia and New Guinea, with the form in the Solomon Islands being *S. s. saccolaimus*.

Milne et al. (2009) demonstrated similarity between the two Australian geographic groups using genetic and morphological analyses. Taxonomic work currently underway, using more powerful nuclear markers, is investigating these groups in the context of the entire species complex (Armstrong pers. comm., cited in Woinarski et al., 2014) and may shed further light on the taxonomic groupings.

For the treatment here, whilst recognising the possibility that current taxonomic studies may conclude differently, we consider that only one taxon occurs in Australia (*S. s. nudicluniatus*), with that taxon also occurring beyond Australia (including New Guinea).

Conservation status

Vulnerable

The bare-rumped sheathtail bat was listed as Critically Endangered under the EPBC Act in 2001. Following a formal review of the listing status of the bare-rumped sheathtail bat, the Threatened Species Scientific Committee (the Committee) has determined that there is sufficient evidence to support a change of status of the subspecies under the EPBC Act from Critically Endangered to Vulnerable.

Species can be listed as threatened under state and territory legislation. For information on the listing status of this species under relevant state or territory legislation, see http://www.environment.gov.au/cgi-bin/sprat/public/sprat.pl.

Reason for conservation assessment by the Threatened Species Scientific Committee

This advice follows assessment of new information provided to the Committee to reassess the listing status of *Saccolaimus saccolaimus nudicluniatus*, for potential removal from the list.

Relevant part of the EPBC Act for amending the list of threatened native species

Section 186 of the EPBC Act states that:

"(2A) The Minister must not delete (whether as a result of a transfer or otherwise) a native species from a particular category unless satisfied that:

(a) the native species is no longer eligible to be included in that category; or

(b) the inclusion of the native species in that category is not contributing, or will not contribute, to the survival of the native species."

Public Consultation

Notice of the proposed amendment and a consultation document was made available for public comment for 32 business days between 29 February 2016 and 15 April 2016. Any comments received that were relevant to the survival of the species were considered by the Committee as part of the assessment process.

Species/Subspecies Information

Description

The bare-rumped sheathtail bat is a large insectivorous bat, with a head and body length of 81−97 mm and a weight of 48−55 g (Hall et al., 2008). It has reddish-brown to dark brown fur on its back and is slightly paler beneath. It can be distinguished from other Australian sheathtail bats (Emballonuridae) by the irregular white flecks of fur on its back and the naked rump (Churchill 1998; Menkhorst & Knight 2001), although not all specimens display these features (Hall et al., 2008). A throat pouch is present in males and is rudimentary in females. Compared to individuals from north-eastern Queensland, those from the Northern Territory may be slightly larger, darker (almost black) on the dorsal fur, with whitish belly fur and lacking the pronounced bare rump (Troughton 1925; McKean et al., 1981; Hall et al., 2008).

Distribution

The bare-rumped sheathtail bat is known to occur in north-eastern Queensland and the monsoonal tropics of the Northern Territory (Milne et al., 2009), and is likely to occur in areas of the Kimberley in Western Australia (Milne pers. comm., cited in Woinarski et al., 2014). In Queensland, it occurs from Ayr to the Iron Range (Dennis 2012), including Magnetic and possibly Prince of Wales Islands (Schulz & Thomson 2007). Most records are near-coastal, but one record (at Jasper Gorge, Northern Territory) has been found 150 km inland (Milne et al., 2009).

There are relatively few records of the subspecies across this extensive range, either suggesting that the subspecies is rare or it has a fragmented distribution. However, issues relating to its detection currently compromise the precise delineation of the subspecies' range and subpopulations: it is morphologically very similar to the yellow-bellied sheathtail bat (*Saccolaimus flaviventris*); is difficult to capture as it mostly flies above the canopy; and its echolocation call pattern is difficult to distinguish from freetail bats and other sheathtail bats within its range.

In 2009, genetic analyses of misidentified specimens of the closely related yellow-bellied sheathtail bat (*Saccolaimus flaviventris*) held at the Northern Territory Museum increased the species' extent of occurrence in the Northern Territory (Milne et al., 2009). In 2011, morphological analyses of four *S. flaviventris* specimens held at the Western Australian Museum indicated that they had been misidentified and are likely to belong to the species *S. saccolaimus* (Milne pers. comm., 2013). The bare-rumped sheathtail bat is therefore likely to be distributed through the Kimberley region of Western Australia as far west as Broome, however this has not been confirmed through genetic analyses (Milne pers. comm., 2013).

Identification of diagnostic characters from full spectrum echolocation recordings has led to further records of the bare-rumped sheathtail bat in new locations in Queensland (Coles et al., 2012). Other potentially useful diagnostic echolocation characters have been reported (Milne et al., 2009; Corben 2010; Ford et al., 2012), but there has not yet been publication of a detailed acoustic comparison of all Australian *Saccolaimus* species (Armstrong pers. comm., cited in Woinarski et al., 2014). If a reliable method for separating them acoustically can be developed, there is potential to better define the range and population size of the bare-rumped sheathtail bat from new surveys and the re-analysis of previous recordings.

Based on the scarcity of records in the previous 16 years, Duncan et al. (1999) considered that the range had probably declined, although were uncertain about their inference: 'it is not clear whether the species [bare-rumped sheath-tail bat] still exists in its former range, or whether the range has changed.' However, given the substantial number of recent records, derived largely from more intensive sampling and better diagnostic capability, there is no substantial evidence of any decline in range.

Relevant Biology/Ecology

In Australia, the bare-rumped sheathtail bat has been recorded mostly in eucalypt forests and woodlands, generally in near-coastal areas. In Queensland, it is known to be associated with coastal lowland rainforests, and more open forests dominated by *Eucalyptus* or *Corymbia* species interspersed with coastal lowland rainforest.

Overseas, the bare-rumped sheathtail bat has been observed roosting in a range of environments, including various hollow-bearing tree species and geological formations, such as caves. However, surveys of caves in Queensland and the Northern Territory have failed to locate this subspecies (Schulz & Thomson 2007). The small number of roosts recorded in Australia have all been found in deep tree hollows of the following species: poplar gum (*Eucalyptus platyphylla*), Darwin woollybutt (*E. miniata*), Darwin stringybark (*E. tetrodonta*) and weeping paperbark (*Melaleuca leucadendra* syn. *leucodendron*) (McKean et al. 1981; Compton & Johnson 1983; Churchill 1998; Murphy 2002; Clague pers. comm. 2013). Hollows in these tree species have also been used as breeding roosts. Such roosts are susceptible to damage by termites and by fire (Churchill 1998; Murphy 2002). Roosts may be used regularly, but individuals may use several roosts, and roost numbers at any site may vary over time (Whybird pers. comm., cited in Woinarski et al., 2014).

The subspecies is insectivorous and forages for flying insects above the canopy (Churchill 1998), although beyond Australia Csorba et al. (2008) considered that it also forages 'close to the ground'. It has been observed foraging within metres of the canopy in riverine gallery forest and *Melaleuca* dominated swamps in Queensland (Clague pers. obs., cited in Woinarski et al., 2014). It is known to fly at altitudes up to and above 400 m and is likely capable of moving long distances (Clague pers. comm. 2015).

Females give birth to a single young, with birth records from Queensland in December and January (Compton & Johnston 1983), and from the Northern Territory from December to about April (Compton & Johnson 1983; Churchill 1998; Milne et al., 2009). Across its global range, the bare-rumped sheathtail bat is considered to be an 'adaptable' subspecies, tolerating some level of disturbance (Csorba et al., 2008).

Generation length is assumed to be 3−5 years, derived from a mean of age at sexual maturity (estimated at 1−2 years) and longevity (probably around 5−8 years), but no detailed information is available for this subspecies.

Threats

Threats to the bare-rumped sheathtail bat are outlined in the table below (Woinarski et al., 2014). Further detail on known and likely threats are in Schulz & Thomson (2007).

Threat factor	Consequence rating	Distributional extent over which threat may operate	Evidence base
Habitat loss and fragmentation	Severe	Localised	The preferred habitat (tall eucalypt open forest) is subject to localised development, mostly for horticulture and urban development (Duncan et al., 1999). The small number of confirmed roosts located in Australia have been in tree hollows; roost sites in trees have been destroyed during clearing (Compton & Johnson 1983).
Competition for tree hollows by bees, non-native and native birds	Minor	Minor	Not demonstrated, but possible (Schulz & Thomson 2007). The spread of the Asian honey bee (*Apis cerana*) in Queensland will increase the competition for hollows in Queensland (Hyatt 2012).
Disease	Unknown	Unknown	Not demonstrated, but possible. Congeners are known to carry the Australian bat Lyssavirus, but the consequences are unknown (Schulz & Thomson 2007; Dennis 2012).
Too frequent burning	Minor	Entire	Not demonstrated, but there are possible impacts on prey abundance and/or availability of large hollow trees used for roosting; its preferred open forest habitat has a very high fire frequency.

How judged by the Committee in relation to the EPBC Act Criteria and Regulations

Criterion 1. Population size reduction (reduction in total numbers) Population reduction (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

Evidence:

Insufficient data to determine eligibility

Previous assessments of the conservation status of the bare-rumped sheathtail bat in Australia have been constrained by taxonomic uncertainty and lack of information about its distribution and range. A study by Milne et al. (2009) has clarified some taxonomic issues, substantially increased the subspecies' known range, and provided more information on its abundance. However, the population size and population trend of the subspecies remain poorly known.

There are relatively few Australian records of the bare-rumped sheathtail bat, especially in Queensland in recent decades (Whybird et al., 2011). However, it is difficult to interpret this meagre information as rarity, as the subspecies is difficult to catch (due to its high flight), and identification was previously constrained by lack of information about call characters that diagnosed it from the more abundant yellow-bellied sheathtail bat *S. flaviventris* (Milne et al., 2009).

Reardon et al. (2010) reviewed the status and distribution records, and undertook additional surveys, for ten microchiropteran bat species on Cape York Peninsula. They noted that most of the priority microbat species on Cape York Peninsula have small and restricted distributions within Cape York Peninsula, and do not appear to face the major threats that typically affect microbats. They further noted that genuine population trends in any species could not be detected, as previous research and monitoring of bats on Cape York Peninsula has been sporadic in time and location.

Habitat loss in some locations can be inferred to have led to, and continue to lead to, some decline in population size which may approach a rate of 10 percent in a three generation period (9−15 years) (Woinarski et al., 2014). However, as the distribution, habitat preferences and biology of the species in Australia remain poorly known (Schulz & Thomson 2007), there is little available information by which to assess whether a decline in distribution or population size may be occurring, or at what rate.

The Committee considers that, based on the information available, it is unlikely that the decline in population size exceeds 50 percent, and the subspecies would probably not meet the eligibility criteria for Endangered or Critically Endangered under this criterion. There is insufficient information to determine the eligibility of the subspecies for listing as Vulnerable under this criterion.

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 km^2**	**< 5,000 km^2**	**< 20,000 km^2**
B2. Area of occupancy (AOO)	**< 10 km^2**	**< 500 km^2**	**< 2,000 km^2**
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 subpopulations; (v) number of mature individuals			
(c) Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations; (iv) number of mature individuals			

Evidence:

Insufficient data to determine eligibility

It is difficult to provide a robust estimate of the current EOO or AOO, as there are few records across its wide distribution, and the number and location of tree-roosts suitable for habitat are likely to vary over time. Based on the mapping of point records from 1976 to 2016, the extent of occurrence is estimated at 1 579 652 km^2, and the area of occupancy estimated at 140 km^2. Point records were 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 (DotE 2015).

Woinarski et al. (2014) considered that the AOO, which they estimated to be 32 km^2, is an under-estimate due to limited sampling across the occupied range, and is likely to be greater than 2000 km^2. The subspecies occurs at more than five locations (Woinarski et al., 2014). A decline in population is inferred from loss of habitat.

The Committee considers that, based on the information available, the AOO is likely to be somewhere between 140 km^2 and 2000 km^2. The subspecies does not meet the eligibility criteria for Endangered or Critically Endangered under this criterion as it occurs at more than 5 locations and no extreme fluctuations are known to occur. However, it may meet the criteria for Vulnerable as the number of locations may be less than 10 and a continuing decline in habitat is inferred, but there is insufficient information to determine this.

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 subpopulation	**≤ 50**	**≤ 250**	**≤ 1,000**
(a) (ii) % of mature individuals in one subpopulation =	**90 – 100%**	**95 – 100%**	**100%**
(b) Extreme fluctuations in the number of mature individuals			

Evidence:

Insufficient data to determine eligibility

There is no robust estimate of population size. Population data are limited as only a small number of roost sites have been found in Australia.

Churchill (1998) noted a record of 40 individuals in one tree roost, and Milne et al. (2009) noted another tree roost containing about 100 individuals. A tree roost noted in Cairns in 2012 contained at least 77 individual bats during peak occupation (Clague pers. comm., cited in Woinarski et al., 2014). A PhD study by Broken-Brow (pers. comm., 2016) found that on Cape York Peninsula known records of the species are limited to 2 or 3 specific locations (despite relatively significant effort across Cape York in the past few years to obtain new records), and the species occurs in extremely low abundance with dusk sightings recording approximately 10 individuals at any one location.

Woinarski et al. (2014) and Armstrong (pers. comm., 2016) suspect the number of mature individuals to be greater than 10 000, given that there is likely to be good roosting potential for the species in a significant proportion of the available habitats across its broad distribution. However, given the limited data available, the number of roost sites and average number of individuals per roost site across the subspecies' distribution cannot be reliably estimated.

The Committee considers that there is insufficient information to determine the eligibility of the species for listing in any category under this criterion.

Criterion 4. Number of mature individuals			
	Critically Endangered Extremely low	**Endangered Very Low**	**Vulnerable Low**
Number of mature individuals	**< 50**	**< 250**	**< 1,000**

Evidence:

Not eligible

Although there is no robust estimate of population size, considering the subspecies' wide distribution, the number of mature individuals is very likely to be greater than 1000 (see also Criterion 3).

The Committee considers that the total number of mature individuals is likely to be greater than 1000 which is not considered extremely low, very low or low. Therefore, the species has not been demonstrated to have met this required element of this criterion.

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**

Evidence:

Insufficient data to determine eligibility

Population viability analysis has not been undertaken.

Consideration for delisting

The bare-rumped sheathtail bat was listed as Critically Endangered under the EPBC Act under Criterion 1 in 2001. The assessment presented in this Conservation Advice suggests that the subspecies may no longer be eligible to be listed under the EPBC Act, as it may not satisfy the listing criteria in any category. New information shows that its range is larger than previously thought, and there is no evidence of a substantial, severe or very severe reduction in population size.

However, the assessment also indicates a deficiency in data for this subspecies. There were insufficient data to assess the subspecies against criteria 1, 2, 3 or 5 to determine whether it meets the eligibility criteria for listing. Assessments against criteria 1 and 2 indicate that the subspecies is unlikely to meet the eligibility criteria for listing as Endangered or Critically Endangered, but may meet the eligilibility criteria for listing as Vulnerable.

The population size and population trends of the subspecies are poorly known, and there are no robust estimates of extent of occurrence or area of occupancy. It is rarely encountered and there are few records; it may be very rare, or more common but rarely reported due to difficulties in low detectability. Considering its habitat requirements, the population may be declining due to habitat quality degradation and habitat loss. Given the uncertainty in the assessment and the suspected population trajectory, there is insufficient evidence to demonstrate that the bare-rumped sheathtail bat should not be included on the threatened species list under the EPBC

Act, as it is possible it may satisfy the criteria for Vulnerable when further information becomes available.

Inclusion of the bare-rumped sheathtail bat in the threatened species list may be contributing to its survival, as the EPBC Act requires project proponents to refer a proposal for assessment if it may have a significant impact on a threatened species/subspecies. Where necessary, the Department has issued conditions requiring proponents to avoid, minimise or mitigate impacts on the bare-rumped sheathtail bat.

Conservation Actions

Recovery Plan

A recovery plan for this species is currently in place. The *National recovery plan for the bare-rumped sheathtail bat* Saccolaimus saccolaimus nudicluniatus (Schulz & Thomson 2007) was developed by the State of Queensland and adopted as a national recovery plan under the EPBC Act in 2008.

The recovery plan includes the following objectives:

- develop more effective detection techniques (including obtaining echolocation reference calls) and undertake systematic surveys to enable a more comprehensive assessment of distribution, population size, status and habitat preferences;
- increase protection of known roosts both on and outside reserved lands;
- better determine roosting requirements and document foraging requirements of the subspecies, including potential seasonal and distributional differences and the identification of threatening processes;
- establish monitoring sites to investigate population trends in the subspecies; and
- clarify the taxonomic status of the subspecies.

Some of these objectives have been achieved, most notably some clarification of its taxonomic status (Milne et al., 2009; Armstrong pers. comm., cited in Woinarski et al., 2012), the characterisation of diagnostic echolocation calls (Clague pers. comm., cited in Woinarski et al., 2012); and more intensive sampling in Cape York Peninsula to improve knowledge of its distribution and status (Reardon et al., 2010). However, no roosts are currently protected from known threatening processes, and habitat critical to survival has not been identified. The plan is scheduled to cease in 2018.

The Committee recommends that the existing recovery plan not be renewed after it ceases in 2018, as its continuation would not add significant benefit above an approved Conservation Advice. This Conservation Advice provides sufficient direction to implement priority actions, mitigate key threats and enable recovery of the subspecies.

Primary Conservation Actions

1. Undertake targeted surveys to identify important subpopulations, roost sites and habitat requirements.
2. Protect important subpopulations, roost sites and mature trees within the subspecies' distribution.
3. Maintain the quality of habitat, particularly around roost sites.
4. Assess trends in population and distribution, and the relative impacts of threats.

Further habitat destruction from activites such as land clearing and mining, in areas containing important subpopulations and roost sites, is likely to have a significant impact on the subspecies. Prior to any clearing or development within the subspecies' distribution, targeted surveys for the bare-rumped sheathtail bat should be undertaken.

Conservation and Management Actions

There are no specific management actions targeted at the bare-rumped sheathtail bat. Parts of its range are included in conservation reserves, where fire management is a priority.

There is no monitoring program specifically for the bare-rumped sheathtail bat. However, there is increased survey and monitoring effort prompted by attempts to resolve the conservation status of poorly-known bat species (e.g. Reardon et al., 2010) and by requirements for environmental impact assessments.

Recent advances in resolving diagnostic features in its echolocation calls have increased the capability to monitor this subspecies using broadband bat detectors, although diagnosis from other *Saccolaimus* species may still not be entirely unambiguous (Armstrong pers. comm., cited in Woinarski et al., 2014). It is not readily caught in harp traps or mist nets set below the canopy. Its use of large trees in forested areas (rather than caves) as roosting sites limits the ability to monitor populations at fixed large roosts. However, if located, roost trees can be monitored by regular stag watches to provide reliable counts of colony size at dusk emergence.

Recommended conservation and management actions are outlined in the table below (Woinarski et al., 2014).

Theme	Specific actions	Priority
Active mitigation of threats	Protect all known roosts and their surrounds within and outside conservation reserves.	High
	Prevent extensive tree clearing in areas occupied by this subspecies; and/or ensure mature trees and corridors are retained.	High
	Reduce the frequency, extent and intensity of fires.	Medium
Captive breeding	N/a	
Quarantining isolated populations	N/a	
Translocation	N/a	
Community engagement	Involve Indigenous ranger groups in survey, monitoring and management.	Medium
	Collaborate with landholders and other stakeholders to prevent loss and disturbance of roost sites.	Medium

Survey and monitoring priorities

Theme	Specific actions	Priority
Survey to better define distribution	Undertake fine-scale sampling to identify and circumscribe important subpopulations (and roost sites), and assess the population size of these.	High
	Undertake broad-scale surveys to assess distribution and abundance.	Medium
Establish or enhance monitoring program	Design an integrated bi-annual monitoring program across its range (including at known roost sites) to determine population trends; surveys should be undertaken in both the wet and dry seasons.	Medium-high
	Implement an integrated monitoring program linked to an assessment of management effectiveness.	Medium-high

Information and research priorities

Theme	Specific actions	Priority
Assess relative impacts of threats	Identify the extent to which suitable roost sites are limiting population size.	Medium
	Identify the population-level responses to a range of fire regimes, and model population viability across all fire scenarios (including consideration of fire impacts on roost site availability).	Medium
	Assess the impact of recently invading insects that may interfere with hollow use (notably Asian honey bees).	Medium
	Assess population-level impacts of clearing on the availability of roost sites.	Low-medium
	Examine patterns of persistence or occurrence in now fragmented habitat.	Low-medium
Assess effectiveness of threat mitigation options	Assess the extent to which tree and/or corridor retention may allow for persistence of this subspecies in modified landscapes.	Medium
	Assess the efficacy and impacts of management options to reduce fire frequency, extent and intensity.	Low-medium
Resolve taxonomic uncertainties	Undertake genetic studies to establish the subspecies' relationships with extralimital forms (Reardon et al. 2010); currently being undertaken by K. Armstrong.	Medium
Assess habitat requirements	Investigate seasonal and spatial patterning of foraging habitat use.	Medium
	Characterise roosting requirements, including maternity and non-breeding roosts.	Medium
Assess diet, life history	Investigate key dietary components	Low-medium
	Assess the extent to which food availability may be affected by fire regimes.	Low-medium

Recommendations

(i) The Committee recommends that the list referred to in section 178 of the EPBC Act be amended by **transferring** from the Critically Endangered category to the Vulnerable category:

Saccolaimus nudicluniatus nudicluniatus

(ii) The Committee recommends there not be a recovery plan for the species.

Threatened Species Scientific Committee

06/09/2016

References cited in the advice

Armstrong, K. N. (2011). The current status of bats in Western Australia. In *The biology and conservation of Australasian bats* (eds B. Law, P. Eby, D. Lunney, and L. Lumsden), pp. 257-269. Royal Zoological Society of New South Wales, Sydney.

Churchill, S. (1998). *Australian Bats*. Reed New Holland, Sydney.

Coles, R. B., Britton, A., Boonman, A. & Clague, C. I. (2012). Discovery of a highly unusual alternating call frequency pattern used by the echolocating emballonurid bat, *Saccolaimus saccolaimus* during foraging. Abstracts from the 15th Australasian Bat Society Conference, Melbourne, 11-13th April 2012. *Australasian Bat Society Newsletter* 38*,* 35.

Compton, A., & Johnston, P. M. (1983). Observations on the sheath-tailed bat, *Taphozous saccolaimus* Temminck (Chiroptera: Emballonuridae), in the Townsville region of Queensland. *Australian Mammalogy 6***,** 83-87.

Corben, C. (2010). Acoustic identification of *Saccolaimus*. Presentation at the 14th Australasian Bat Society conference, Darwin, Northern Territory, 12–14 July 2010.

Corbet, G. B., & Hill, J. E. (1980). *A world list of mammalian species*. British Museum (Natural History), London.

Csorba, G., Bumrungsri, S., Francis, C., Helgen, Bates, P., Heaney, L., Balete, & Thomson, B. (2008). *Saccolaimus saccolaimus*. In 'The IUCN Red List of Threatened Species.' Viewed 22 May 2012. Available on the internet at: http://www.iucnredlist.org.

Dennis, A. J. (2012). Bare-rumped sheathtail bat, *Saccolaimus saccolaimus nudicluniatus*. In *Queensland's threatened animals* (eds L. K. Curtis, A. J. Dennis, K. R. McDonald, P. M. Kyne and S. J. S. Debus), p. 396. CSIRO Publishing, Collingwood.

Department of the Environment (DotE) (2015). *Area of Occupancy and Extent of Occurrence for Saccolaimus saccolaimus nudicluniatus*. Unpublished report, Australian Government Department of the Environment, Canberra.

De Vis, C. W. (1905). Bats. *Annals of the Queensland Museum 6*, 36-40.

Duncan, A., Baker, G. B., & Montgomery, N. (1999). *The action plan for Australian bats*. Environment Australia, Canberra.

Flannery, T. F. (1990). *Mammals of New Guinea.* Robert Brown & Associates, Carina.

Flannery, T. F. (1995). *Mammals of the South-west Pacific and Moluccan islands*. Reed, Sydney.

Ford, G., Thomson, B., & Macdonald, S. (2012). The ongoing challenge of characterising the echolocation calls of a rare tropical bat. Presentation at the 15th Australasian Bat Society Conference, Melbourne, Australia, 11-13 April 2012.

Goodwin, R. E. (1979). The bats of Timor: systematics and ecology. *Bulletin of the American Museum of Natural History 163*, 72-122.

Hall, L. S., Thomson, B. G., & Milne, D .J. (2008). Bare-rumped Sheath-tailed Bat *Saccolaimus saccolaimus*. In *Mammals of Australia*. Third edition. (Eds S. van Dyck & R. Strahan), pp. 475-476. Reed New Holland, Sydney.

Hyatt, S. (2012). Asian Honey Bee (*Apis cerana javana*) in Cairns, Far North Queensland: foraging, nesting and swarming behaviour. Report of field observations April 2007-September 2011. Department of Employment, Economic Development and Innovation, Brisbane.

Koopman, K. (1984). Taxonomic and distributional notes on tropical Australian bats. *American Museum Novitates 2778*, 1-48.

Koopman, K. F. (1994). Chiroptera: systematics. In *Handbook of Zoology*. Volume 8, Part 60. Walter de Gruyter, Berlin.

McKean, J. L., Friend, G., & Hertog, A. L. (1981). Occurrence of the sheath-tailed bat *Taphozous saccolaimus* in the Northern Territory. *Northern Territory Naturalist 4*, 20.

Menkhorst, P. & Knight, F. (2001). *A field guide to the Mammals of Australia*. Oxford University Press, Victoria.

Milne, D. J., & Pavey, C. R. (2011). The status and conservation of bats in the Northern Territory. In *The biology and conservation of Australasian bats* (eds B. Law, P. Eby, D. Lunney, & L. Lumsden), pp. 208-225. Royal Zoological Society of New South Wales, Sydney.

Milne, D. J., Jackling, F. C., Sidhu, M., & Appleton, B. J. (2009). Shedding new light on old species identifications: morphological and genetic evidence suggest a need for conservation status review of the critically endangered bat *Saccolaimus saccolaimus*. *Wildlife Research 36*, 496-508.

Murphy, S. 2002. Observations of the 'Critically Endangered' Bare-rumped Sheathtail Bat *Saccolaimus saccolaimus* Temminck (Chiroptera: Emballonuridae) on Cape York Peninsula, Queensland. *Australian Mammalogy 23*, 185-187.

Nowak, R. M., & Paradiso, J. L. (1983). *Walker's Mammals of the World*. Johns Hopkins University Press, Baltimore.

Reardon, T. B., Robson, S. K. A., Parsons, J. G., & Inkster, T. (2010). Review of the threatened status of microchiropteran bat species on Cape York Peninsula. South Australian Museum, Adelaide.

Schulz, M., & Thomson, B. (2007). *National recovery plan for the bare-rumped sheathtail bat* Saccolaimus saccolaimus nudicluniatus*.* Queensland Environmental Protection Agency, Brisbane.

Simmons, N. B. (2005). Order Chiroptera. In *Mammal species of the world: a taxonomic and geographic reference*. Third edition. (Eds. D. E. Wilson & D. M. Reeder), pp. 312–529. Johns Hopkins University Press, Baltimore.

Thomson, B. (1991). *A field guide to bats of the Northern Territory*. Conservation Commission of the Northern Territory, Darwin.

Troughton, E. Le G. (1925). A revision of the genera *Taphozous* and *Saccolaimus* (Chiroptera) in Australia and New Guinea, including a new species, and a note on two Malayan forms. *Records of the Australian Museum 14*, 313-341.

Whybird, O. J., Tully, P. J., Clague, C. I., & Coles, R. B. (2011). Live report of the critically endangered Bare-rumped Sheathtail Bat (*Saccolaimus saccolaimus*) in north Queensland, Australia. *Australasian Bat Society Newsletter 37*, 20-21.

<u>Other sources cited in the advice</u>

Clague, C. (2015). Personal communication by email, 7 March 2015. Queensland Department of Agriculture and Fisheries.

Armstrong, K. (2016). Comment on the conservation listing re-assessments for *Hipposideros semoni*, *Rhinolophus robertsi* and *Saccolaimus saccolaimus*. Personal communication by email, 15 April 2016. Zoologist, member of the IUCN Bat Specialist Group.

Broken-Brow, J. (2016). Comment on listing re-assessment of *Saccolaimus saccolaimus*. Personal communication by email, 15 April 2016. PhD candidate, University of Queensland.

Southern Black-throated Finch (Poephila cincta cincta)

Advice to the Minister for the Environment and Heritage from the Threatened Species Scientific Committee (TSSC) on Amendments to the list of Threatened Species under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act)
14 February 2005

1. Scientific name, common name (where appropriate), major taxon group.

Poephila cincta cincta (Southern Black-throated Finch)

Two subspecies of Black-throated Finch are recognised, the northern subspecies (*Poephila cincta atropygialis*) and the southern subspecies (*Poephila cincta cincta*). The southern subspecies of the Black-throated Finch currently occurs in coastal northern Queensland and inland central Queensland. The northern subspecies occurs in northern Queensland from the Atherton Tablelands in the south to the Cape York Peninsula in the north and to the Gulf of Carpentaria in the west. Historically there was a broad region of interbreeding and intergradation between the northern and southern subspecies along the Burdekin-Lynd Divide in a broad band west-southwest.

2. Description

The southern Black-throated Finch is a small stocky granivorous bird. With pink feet, dark grey bill, cinnamon breast, grey head, brown back, black tail, white rump and black throat, the southern Black-throated Finch or Parson's Finch, is a particularly striking looking bird popular with aviculturalists. The southern subspecies is distinguished from the northern subspecies by its colouring. The southern subspecies has a white rather than black rump and richer brown plumage. The female has a slightly smaller black throat patch than the male.

The southern Black-throated Finch occupies woodland savannah and riverine vegetation. Inland it prefers grassy woodland dominated by eucalypts, paperbacks or acacias, where there is access to seeding grasses and water (Zann 1976). On the coast, it occupies open grassy plains with Pandanus (Pizzey 1991).

The subspecies feeds mainly on half-ripe seeds of various grasses that have fallen to the ground, but will also eat insects such as flying insects, spiders and ants and their larvae (Zann 1976).

3. National context

The southern subspecies of the Black-throated Finch currently occurs in coastal northern Queensland and inland central Queensland. It occurs in the northern end of the Brigalow belt and west into the Einasleigh Uplands. It remains locally common at only a few sites near Townsville and Charters Towers (NSW and Queensland Governments 2004).

The range of the southern subspecies of the Black-throated Finch has contracted considerably. Historically, the southern subspecies of the Black-throated Finch had a wider distribution occurring further south in central and southeast Queensland through to northeast New South Wales.

There is some anecdotal evidence that the northern subspecies may have also declined in range and abundance but there is no scientific evidence to suggest that it has declined to any substantial degree (Franklin 1999).

Black-throated Finches are also popular with aviculturists and there are large numbers bred in captivity.

The southern Black-throated Finch (*Poephila cincta cincta*) is currently listed as Vulnerable under the *Environment Protection and Biodiversity Conservation Act 1999*. It is also currently listed as endangered under the NSW *Threatened Species Conservation Act 1995* and as vulnerable under the Queensland *Nature Conservation Act 1992*. The southern Black-throated Finch is listed under the Convention on International Trade in Endangered Fauna and Flora (CITES) Appendix II.

EPBC Act criteria.

TSSC judges Poephila cincta cincta to be eligible for listing as endangered under the EPBC Act. The justification against the criteria is as follows:

Criterion 1 - It has undergone, is suspected to have undergone or is likely to undergo in the immediate future a very severe, severe or substantial reduction in numbers.

The southern Black-throated Finch historically had a wide distribution occurring from Queensland's Atherton Tablelands in the north to northeast New South Wales in the south and to central Queensland in the west. The southern Black-throated Finch's range has considerably contracted and is now only found to occur in coastal northern Queensland and inland central Queensland. It is suspected to undergone a severe reduction in numbers as a result of this severe decline in range.

The southern Black-throated Finch has been reasonably well surveyed especially at the southern most end of its former range. The most recent confirmed sighting of the southern Black-throated Finch in the New England Tablelands area of NSW was in 1994. As it was suspected that the subspecies had become locally extinct since, in 2000 Birds Australia conducted a systematic survey of locations with suitable habitat and areas where it had previously been sighted. Despite an intensive search of their former range, no Black-throated Finches were sighted (Ley and Cook 2001).

The most recent confirmed sighting of the southern Black-throated Finch in southern Queensland was in Stanthorpe in 1995. While there has been some anecdotal evidence that the subspecies may continue to occur in this area, there is no scientific data to support this. Although there has been no species-specific survey for the southern Black-throated Finch in southern Queensland, the Australian Bird Atlas of 1998-2001 systematically surveyed this area for all bird species and no Black-throated Finches were sighted.

The comparative analysis of the two Bird Atlases also showed a marked decline in reporting rates for southern Black-throated Finches over the past 20 years for all IBRA bioregions in which the subspecies was formerly found. Although there is debate about the validity of such comparisons and the analysis is considered inconclusive, the indicative trend from the reporting rate combined with the distribution data infer that there is a corresponding reduction in population size of a similar magnitude to the decline in range.

The extent of occurrence of the southern Black-throated Finch has declined considerably in the last 50 years or more. The extent of occurrence in the 1977-1981 period using the first Bird Atlas data has been estimated as 112,000- 311,000 km^2 depending on whether a single large polygon is drawn around the records to show the full historic range or two smaller polygons are drawn around the two remaining strongholds. The current extent of occurrence using the second Bird Atlas (1998-2003) has been estimated as 52,000 km^2 from this latter data. In total the subspecies has undergone a severe decline of 53-83% over two decades. Assuming a steady rate of decline, this is equivalent to a 32-59% decline in the last 10 years.

There has been some debate about the estimated extent of occurrence. Although the current extent of occurrence has been calculated using Bird Atlas records only, it is considered to be reasonably accurate as other data from this time period including the Black-throated Finch Recovery Team's database and Queensland's Environmental Protection Agency records also fall within this area.

Historically, it has been observed that the two subspecies of Black-throated Finch cohabitate and interbreed over a broad band of intergradation (Ford 1986, Schodde and Masson 1999). One study (Zann 1976) however suggested that the zone of interbreeding had moved further south, that the northern subspecies was pushing into the southern subspecies range and that there was little actual interbreeding in this zone but more co-occurrence of the two subspecies. There have been no recent observations of the southern subspecies in the southern part of this zone and only the northern subspecies has been observed in the northern part of the zone (NSW and Queensland Governments 2004). This suggests there is now little overlap in the range of the northern and southern subspecies. Even if the southern subspecies does still occur within this band of intergradation, the major contraction in the extent of occurrence is at the southern end of the Black-throated Finch's range and including the zone of intergradation in the estimated extent of occurrence is unlikely to lessen the severity of the decline.

The decline of the southern Black-throated Finch and many other granivorous bird species appears to have begun with the introduction of pastoralism in the early 20th Century (Franklin 1999). The loss or degradation of its preferred riparian grassland habitat through inappropriate fire regimes, clearing for agriculture or development, spread of introduced grasses, and overgrazing particularly in combination with drought has had significant impacts on this subspecies.

A study of the bird fauna (including the Black-throated Finch) of the Coomooboolaroo property in central Queensland documented some of these threats and the resulting decline in avifauna (Woinarski and Caterrall 2004). A number of environmental changes appear to have occurred concurrently, the individual effects of which are unknown but together have caused significant change in the landscape. These include: fire exclusion/cessation of aboriginal fire management and resulting woody regrowth; increased stocking rates; drought; episodic mammal 'plagues' (dingoes, marsupials and feral cats) and subsequent baiting; ring-barking of trees; clearing of brigalow 'scrub' and the subsequent introduction of exotic pasture grasses; proliferation of the introduced prickly pear cactus and the cane toad; and the degradation of watercourses through trampling and subsequent sedimentation. 45% of the bird species found in 1873, including the Black-throated Finch, declined or became locally extinct by 1999. The study found that the negative effects of these changes were more likely to affect small-bodied birds, particularly those dependent on grassy forest/woodland habitats. The study also suggests that the same processes that occurred at Coomooboolaroo are likely to be widespread, occurring over the northern Australian savannas as a whole.

As this is a highly sought after species for aviculturalists, it is thought that illegal trapping may also have contributed to its decline causing the local extinction of some remnant populations (Garnett and Crowley 2000).

The southern subspecies of the Black-throated Finch is suspected to have undergone a severe reduction in numbers as there has been an observed decline in extent of occurrence of up to 59% over the last decade due mainly to the spread of pastoralism and associated changes in land management practices. Therefore, the species is **eligible** for listing as **endangered** under this criterion.

Criterion 2 - Its geographic distribution is precarious for the survival of the species and is very restricted, restricted or limited.

The current extent of occurrence of the southern Black-throated Finch species has been estimated as 52,000 km^2. Although, the southern Black-throated Finch's range is declining, at this stage its geographic distribution is not considered limited or precarious for the survival of the species.

As the southern Black-throated Finch population has declined, a number of geographically separated, remnant subpopulations probably formed including an isolated subpopulation in northeast NSW and southeast Queensland. Despite recent survey efforts, these subpopulations appear to be locally extinct with the subspecies now occurring in northern Queensland only. It is not known at how many locations the subspecies remains.

The southern Black-throated Finch is thought to be sedentary, but erratic appearance at certain localities suggests there may be some local movement possibly in response to drought (Ley & Cook 2000). There does not however appear to be extreme fluctuations in the extent of occurrence or number of locations.

The greatest known threat to the southern Black-throated Finch appears to be loss or degradation of its preferred riparian grassland habitat due mainly to the spread of pastoralism and associated changes in land management practices (see Criterion 1). Illegal trapping may also be a threat.

Although the species has been subject to a decline and threats are continuing, the current extent of occurrence is not considered limited and precarious for the survival of the species and therefore, the species is **not eligible** for listing under this criterion.

Criterion 3 - The estimated total number of mature individuals is limited to a particular degree and: (a) evidence suggests that the number will continue to decline at a particular rate; or (b) the

number is likely to continue to decline and its geographic distribution is precarious for its survival.

The only available population estimate is 20,000 (Garnett and Crowley 2000) but it is unknown how this figure was estimated and it is thought to be of low reliability. There is also no quantitative data on actual declines in population.

The comparative analysis of the reporting rates for southern Black-throated Finches between the two Bird Atlases indicates a pattern of population decline. There is a marked decline in reporting rates over the past 20 years for all IBRA bioregions in which the subspecies was formerly found. The use of reporting rates to estimate actual declines however is problematic as survey methods differed between the two Atlases and there were no allowances for seasonal differences. This analysis is considered inconclusive and mainly indicative.

As there is no quantitative data available against this criterion, therefore, the species is **not eligible** for listing under this criterion.

Criterion 4 - The estimated total number of mature individuals is extremely low, very low or low.

The estimated population is 20,000 (Garnett and Crowley 2000). Therefore, the species is **not eligible** for listing under this criterion.

Criterion 5 - Probability of extinction in the wild

There is no quantitative data available against this criterion. Therefore, the species is **not eligible** for listing under this criterion.

5. Conclusion

The southern Black-throated Finch is suspected to have undergone a severe reduction in numbers, as there has been an observed decline in extent of occurrence of up to 59% in the last 10 years. The subspecies historically had a wide distribution occurring from Queensland's Atherton Tablelands in the north to northeast New South Wales in the south and to central Queensland in the west. The southern Black-throated Finch is now only found to occur in coastal northern Queensland and inland central Queensland.

The greatest known threat to the southern Black-throated Finch appears to be loss or degradation of its preferred riparian grassland habitat due mainly to the spread of pastoralism and associated changes in land management practices.

The species is **eligible** for listing as **endangered** under criterion 1.

6. Recommendation

TSSC recommends that the list referred to in section 178 of the EPBC Act be amended by **transferring from the vulnerable category** to the **endangered category**:

Poephila cincta cincta (Southern Black-throated Finch)

Publications used to assess the nomination

Barrett G., A. Silcocks, S. Barry, R. Cunningham and R. Poulter (2003). The New Atlas of Australian Birds, CSIRO Publishing.

Blakers M., S.J.J.F. Davies & P.N. Reilly (1984). The Atlas of Australian Birds, Melbourne University Press.

Ford, J. (1986). Avian hybridisation and allopatry in the region of the Einasleigh Uplands and Burdekin-Lynd Divide, north-eastern Queensland *in* Emu 86: 87-110

Franklin D.C. (1999). Evidence of disarray amongst granivorous bird assemblages in the savannas of northern Australia, a region of sparse human settlement *in* Biological Conservation 90: 53-68

Garnett S.T. & G.M. Crowley (2000). The Action Plan for Australian Birds 2000

Ley A. & S. Cook (2001). The Black-throated Finch *Poephila cincta* in New South Wales in Australian Bird Watcher 19:115-120

NSW and Queensland Governments (2004). Draft Recovery Plan for the Black-throated Finch southern subspecies *Poephila cincta cincta*. NSW Department of Environment and Conservation, Queensland Environmental Protection Agency and Queensland Parks and Wildlife Service.

Pizzey G. (1991). A Field Guide to the Australian Birds. Revised Edition. Angus and Robertson, Sydney.

Schodde R. & I.J. Mason (1999). The Directory of Australian Birds: Passerines.

Woinarski J.C.Z. and C.P. Caterrall (2004). Historical changes in bird fauna at Coomooboolaroo, northeastern Australia, from the early years of pastoral settlements (1873) to 1999 *in* Biological Conservation. 116: 379-401.

Zann R. (1976). Distribution, status and breeding of Black-throated Finches *Poephila cincta* in northern Queensland *in* Emu 76: 201-206

Conservation Advice

The southern Black-throated Finch is a small, striking looking, granivorous bird which occupies woodland savannah and riverine vegetation in coastal northern Queensland and inland central Queensland (NHT regions). The subspecies occurs predominantly on leasehold, freehold and council land. The key threat to this subspecies is the loss or degradation of habitat due to changes in land use management practices.

The priority recovery and threat abatement actions required for this species are:

- Protecting and enhancing habitat where the species is know to occur including securing sites for conservation, involving land mangers in conservation, and monitoring management effectiveness.

This list does not encompass all actions that may be of benefit to this species, but highlights those that are considered to be of the highest priority at the time of listing.

The Queensland and NSW Governments have developed a joint draft Recovery which sets out specific management actions and guidelines for the conservation of this subspecies.

Priority for the development of recovery plan: N/A

Conservation Advice

Calidris ferruginea

curlew sandpiper

Taxonomy

Conventionally accepted as curlew sandpiper *Calidris ferruginea* Pontoppidan, 1763. Scolopacidae. Other common names are pygmy curlew, curlew stint and redcrop.

No subspecies are recognised (Bamford et al. 2008). Taxonomic uniqueness: medium (22 genera/family, 20 species/genus, 1 subspecies/species; Garnett et al. 2011).

Cox's sandpiper (*Calidris paramelanotos*) was described as a new species in 1982, but is now known to be a hybrid between a female curlew sandpiper and a pectoral sandpiper (*C. melanotos*) (McCarthy 2006; Christidis & Boles 2008). Before 1990 there were said to be 4-7 (unverified) Australian reports of Cox's sandpiper annually (Higgins & Davies 1996), but reports are now very rare. Curlew sandpipers have also been reported to hybridise with white-rumped sandpipers (*Calidris fuscicollis*) (McCarthy 2006).

Summary of assessment

Conservation status

Critically endangered: Criterion 1 A2, (a)

Calidris ferruginea has been found to be eligible for listing under the following listing categories:

Criterion 1: A2 (a): Critically Endangered
Criterion 2: Not eligible
Criterion 3: Not eligible
Criterion 4: Not eligible
Criterion 5: Not eligible

The highest category for which *Calidris ferruginea* is eligible to be listed is Critically Endangered.

Species can be listed as threatened under state and territory legislation. For information on the listing status of this species under relevant state or territory legislation, see http://www.environment.gov.au/cgi-bin/sprat/public/sprat.pl

Reason for conservation assessment by the Threatened Species Scientific Committee

This advice follows assessment of information provided by a committee nomination based on information provided in the *Action Plan for Australian Birds 2010* (Garnett et al., 2011), and experts from the University of Queensland.

Public Consultation

Notice of the proposed amendment and a consultation document was made available for public comment for 33 business days between 1 October 2014 and 14 November 2014. Any comments received that were relevant to the survival of the species were considered by the Committee as part of the assessment process.

Species Information

Description

The curlew sandpiper is a small, slim sandpiper 18–23 cm long and weighing 57 g, with a wingspan of 38–41 cm. It has a long decurved black bill with a slender tip; the legs and neck are also long. The head is small and round, and the iris is dark brown. The legs and feet are black or black-grey. When at rest, the wing-tips project beyond the tip of the tail. It has a square white patch across the lower rump and uppertail-coverts, a prominent flight character in all plumages. The sexes are similar, but females have a slightly larger and longer bill and a slightly paler underbelly in breeding plumage (Higgins & Davies, 1996).

In breeding plumage, the head, neck and underbody to rear belly are a rich chestnut-red with narrow black bars on the belly and flanks. There are black streaks on the crown, a dusky loral stripe, and white around the base of the bill. When the plumage is fresh, the head, neck and underbody are often mottled by white tips to the feathers. The feathers on the mantle and scapulars are black with large chestnut spots and greyish-white tips (Higgins & Davies, 1996).

The non-breeding plumage looks very different, with pale brownish grey upperparts and predominantly white underparts (with a brownish-grey wash and fine dark streaks on the foreneck and breast). The cap, ear-coverts, hindneck and sides of neck are pale brownish-grey with fine dark streaks, grading to off-white on the lower face, with white on the chin and throat. There is a narrow dark loral stripe and white supercilium from the bill to above the rear ear-coverts. (Higgins & Davies, 1996).

Distribution

Australian distribution

In Australia, curlew sandpipers occur around the coasts and are also widespread inland, though erratic in their appearance across much of the interior. There are records from all states during the non-breeding period, and also during the breeding season when many non-breeding birds remain in Australia rather than migrating north.

In Queensland, scattered records occur in the Gulf of Carpentaria, with widespread records along the coast south of Cairns. There are sparsely scattered records inland. In NSW, they are widespread east of the Great Divide, especially in coastal regions. They are occasionally recorded in the Tablelands and are widespread in the Riverina and south-west NSW, with scattered records elsewhere. In Victoria, they were widespread in coastal bays and inlets; despite recent declines these are still their Victorian strongholds; they are widespread in near-coastal wetlands, and they occur intermittently on inland wetlands (e.g. in the Kerang area, Mildura, and western districts). In Tasmania, they were recorded on King Island and the Furneaux Group. They mostly occur in south-eastern Tasmania, but also at several sites in north-west Tasmania, with occasional records in low numbers on the west coast. In South Australia, curlew sandpipers occur in widespread coastal and sub-coastal areas east of Streaky Bay. Important sites include ICI and Price Saltfields, and the Coorong. Occasionally they occur in inland areas south of the Murray River and elsewhere. In Western Australia, they are widespread around coastal and sub-coastal plains from Cape Arid to south-west Kimberley. They occur in large numbers, in thousands to tens of thousands, at Port Hedland Saltworks, Eighty-mile Beach, Roebuck Bay and Lake Macleod. They are rarely recorded in the north-west Kimberley, around Wyndham and Lake Argyle, and occasionally they occur inland, in areas south of 26° S. In the Northern Territory, they mostly occur around Darwin, north to Melville Island and Cobourg Peninsula, and east and south-east to Gove Peninsula, Groote Eylandt and Sir Edward Pellew Island. They have been recorded inland from Victoria River Downs and around Alice Springs (Higgins & Davies, 1996).

Global distribution

The global population size of the curlew sandpiper has been estimated to be 1,350,000 (Delany & Scott, 2002; Bamford et al., 2008), however, these estimates are out of date. The global extent of occurrence is estimated at 100 000–1 000 000 km² (BirdLife International, 2014). Approximately 13% of the global population occurs in the East Asian-Australasian Flyway (180

000 individuals) (Bamford et al., 2008), however, these estimates are out of date and the true estimate is probably much lower.

The breeding range of the curlew sandpiper is restricted to the Russian Arctic from Chosha Bay east to Kolyuchiskaya Bay, on the Chukchi Peninsula, and also the New Siberian Islands (Lappo et al., 2012). It is a passage migrant through Europe, north Africa, Kazakhstan, west and south-central Siberia, Ussuriland, China, Taiwan, Japan, the Philippines and Papua New Guinea.

During the non-breeding period, they occur throughout Africa, south of southern Mauritania and Ethiopia, along the valley of the Nile River and in Madagascar. They also occur in Asia, from the coastal Arabian Peninsula to Pakistan and India, through Indonesia and Malaysia, south-east Asia and Indochina to south China and Australasia (Higgins & Davies, 1996).

Relevant Biology/Ecology

Life history

A generation time of 7.6 years (BirdLife International, 2014) is derived from an age at first breeding of 2.0 years, an annual survival of adults of 79% and a maximum longevity of 14.8 years, all extrapolated from congeners (Garnett et al., 2011). Estimates of apparent and true survival rate respectively for curlew sandpipers in Victoria are 73.1% and 80.5% (Rogers and Gosbell 2006). Rogers and Gosbell (2005) demonstrated that long-term decline in Victorian curlew sandpipers, although influenced by consecutive years of low breeding success, has been driven by reduced adult survival. Minton et al. (2006) confirmed that curlew sandpipers do not begin northwards migration and breeding until 2 years old.

Data extracted from the Australian Bird and Bat Banding Scheme (ABBBS) reports a longevity record of 18 years, 1.9 months (Australian Government, 2014).

Breeding

This species does not breed in Australia.

In Siberia, nesting occurs during June and July (Hayman et al., 1986). The nest is a cup positioned on the margins of marshes or pools, on the slopes of hummock tundra, or on dry patches in *Polygonum* tundra (BirdLife International, 2014). Curlew sandpipers usually have a clutch size of four eggs (Johnsgard, 1981).

General habitat

In Australia, curlew sandpipers mainly occur on intertidal mudflats in sheltered coastal areas, such as estuaries, bays, inlets and lagoons, and also around non-tidal swamps, lakes and lagoons near the coast, and ponds in saltworks and sewage farms. They are also recorded inland, though less often, including around ephemeral and permanent lakes, dams, waterholes and bore drains, usually with bare edges of mud or sand. They occur in both fresh and brackish waters. Occasionally they are recorded around floodwaters (Higgins & Davies, 1996).

"*The Shorebird Community occurring on the relict tidal delta sands at Taren Point*" is listed as an Endangered Ecological Community in NSW (NSW DECC, 2005). The curlew sandpiper is one of 20 shorebird species that make up this community.

Feeding habitat

Curlew sandpipers forage on mudflats and nearby shallow water. In non-tidal wetlands, they usually wade, mostly in water 15–30 mm, but up to 60 mm deep. They forage at the edges of shallow pools and drains of intertidal mudflats and sandy shores. At high tide, they sometimes forage among low sparse emergent vegetation, such as saltmarsh, and sometimes forage in flooded paddocks or inundated saltflats. Occasionally they forage on wet mats of algae or waterweed, or on banks of beachcast seagrass or seaweed. They rarely forage on exposed reefs (Higgins & Davies, 1996). In Roebuck Bay, northern Western Australia, they tend to follow the receding tide to forage near the water edge (Rogers 1999, 2005) but they also feed on part of the mudflats that have been exposed for a longer period, foraging in small groups (Tulp & de Goeij, 1994).

Roosting habitat

Curlew sandpipers roost in open situations with damp substrate, especially on bare shingle, shell or sand beaches, sandspits and islets in or around coastal or near-coastal lagoons and other wetlands, occasionally roosting in dunes during very high tides and sometimes in saltmarsh (Higgins & Davies, 1996). They have also been recorded roosting in mangroves in Inverloch, Victoria (Minton & Whitelaw, 2000).

Feeding

This species forages mainly on invertebrates, including worms, molluscs, crustaceans, and insects, as well as seeds. Outside Australia, they also forage on shrimp, crabs and small fish. Curlew sandpipers usually forage in water, near the shore or on bare wet mud at the edge of wetlands. On wet mud they forage by pecking and probing. They probe in shallow water, and jab at the edge of the water where a film of water remains on the sand. They glean from mud and less commonly from the surface of water, or in drier areas above the edge of the water. For a 'jab' less than half the length of the bill is inserted into the substrate; a probe is performed with a slightly open bill inserted to its full length. Curlew sandpipers may wade up to the belly, often with their heads submerged while probing. They often forage in mixed flocks (Dann, 1999a), including with red-necked stints (*Calidris ruficollis*).

The diet of the curlew sandpiper includes the following taxa (Barker & Vestjens, 1989; Higgins & Davies, 1996; Dann, 1999a):

Plants (*Ruppia* spp. seeds), Annelid worms: *Ceratonereis eurythraeensis*, *Nereis caudate,* Molluscs: Kelliidae, Gastropods: Rissoidae, Cerithiidae, Fossaridae, *Polinices* sp., *Salinator fragilis*, Hydrococcidae, Hydrobiidae, *Assiminea brazieri*, *A. tasmanica*, Crustaceans: *Cymadusa* sp., *Paracorophium* sp., Brachyurans; Sentinel Crab (*Macrophthalamus latifrons*), Insects: Diptera (Stratiomyidae, Chironomidae), adults, larvae and pupae, larvae (of Coleoptera, Dytiscidae and Scarabaeidae), Lepidoptera

Curlew sandpipers have been recorded consuming grit. In tidal waters, on the outgoing tide, the birds move onto the most recently exposed parts of the tidal flats until low tide when they disperse widely (Rogers 1999). On the rising tide, the flocks remain in areas close to the water's edge until these areas are covered and then retreat in stages rather than moving continuously as they do on the outgoing tide. Occasionally, individuals feed at high tide near the roost, along stretches of sandy beach where piles of decomposing vegetation are scattered in the high-tide zone. Supratidal feeding mainly occurs during the pre-migratory fattening periods (February-April) (Dann, 1999b). In other studies supratidal foraging has been recorded throughout the austral summer, and has been found to occur more on neap tides when tidal flat exposure is reduced (Rogers et al. 2013).

Migration patterns

Curlew sandpipers are migratory. Overlapping breeding grounds occur in Siberia, and populations move south to widely different non-breeding areas which generally occur south of 35° N. Most birds migrate south, probably overland across Siberia and China, and south Asia. The northern migration occurs much further east, mainly along the south-east and east coasts of China, where staging occurs, then continuing overland to breeding areas (Higgins & Davies, 1996).

Departure from breeding grounds

Males depart breeding grounds during early July, followed by females in July and early August, then juveniles in August, with juveniles usually arriving in the non-breeding range later than adults. Southwards migration is poorly known but flag resightings indicate that the main passage is initially overland, and that some birds migrate well to the west of the direct great circle route from the breeding grounds to south-eastern Australia (Minton et al., 2006). They cross Russia during July till late October, and pass through Mongolia, with a few records from inland Asia. They reach the Asian coast on a broad front between India and China in August. Adults pass through the Inner Gulf of Thailand during August, with a second influx, probably mainly juveniles, in late October and early November. Thousands pass over the west coast of Malaysia and arrive in Singapore in July and August but the migratory destination of these birds is unclear. Small numbers pass through Myanmar and Hong Kong during August-October. The relatively low numbers of curlew sandpipers, and of resightings of Australian-flagged birds on the coast of Indonesia, Borneo, the Philippines and Papua New Guinea, suggest that curlew sandpipers migrating to Australia migrate in a direct flight from staging areas on the east Asian coast. They are regular in small numbers on passage through southern Papua New Guinea, and in the Port Moresby district they arrive as early as late August. Adults are capable of flying non-stop to Australia from Hong Kong and Singapore. They reach the northern shores of Australia in late August and early September (Higgins & Davies, 1996; Minton, 1996; Minton et al., 2006).

Non-breeding season

Substantial numbers of Curlew Sandpipers remain in northern Australia throughout the non-breeding season (e.g. Rogers et al. 2008). Others stopover in northern Australia before continuing migration to south-east Australia, the first birds arriving in late August, but the majority not until September. Some birds are also thought to move through the Gulf of Carpentaria to east and south-east Australia, with records from coastal Queensland and NSW. Some, occasionally hundreds, pass through north-east South Australia during late August to early December, and small numbers occur regularly in south-west NSW from early August. Some birds also move from north-west Australia, south to southern Western Australia, sometimes arriving in coastal south-western Western Australia as early as August, with small numbers also passing through Eyre, south-eastern Western Australia, mainly during August-November. Birds may return to the same non-breeding sites each year (Higgins & Davies, 1996; Minton, 1996).

Return to breeding grounds

The return north begins in March, the northern route being further to the east than the southern route. Sightings of colour-marked birds, and influx at inland sites in south-eastern Australia in April, suggest some passage occurs through inland areas, and at least some birds from south-eastern Australia move to north-west Australia before leaving the mainland. Curlew sandpipers leave coastal sites in east Queensland between mid-January and mid-April, with a possible passage along the north-east coast. They migrate north on a broad front, with fewer occurring in north-west Australia than on the southern migration. Young birds stay in non-breeding areas during breeding season (Higgins & Davies, 1996). Recoveries and flag resightings indicate that a large proportion of the Australian population migrate through southern China (including Hong Kong and Taiwan), Vietnam and Thailand in the last few days of March and through April. Migration is however on a broad front and smaller numbers of birds pass though Papua New Guinea in early April to mid-May, and Bali and Sumatra during March-April. Small numbers pass through Brunei, during mid-February to May, with large numbers passing through the Philippines during March-April. The birds depart Singapore during early March, passing through Malaysia during March-April. They move through the Inner Gulf of Thailand during late March-May and depart Myanmar during May. By May the majority of recoveries and flag resightings occur on or near the Asian coast, notably on the northern coast of Bohai Bay, with other major concentrations in the Yangtse Estuary and the northern base of the Shandong Peninsula. A few pass through the Republic of Korea, Japan and Sakhalin during April-May. They first arrive in Chukotka region, Russia, during late in May or early June (Higgins & Davies, 1996; Minton, 1996, Minton et al. 2006, Hong-Yan et al. 2011).

Descriptions of migratory pathways and important sites

Birds banded in Australia have been recovered in the upper Yenisey River and Daursky Nature Reserve, Russia, south India, Tanggu near Tianjin, many in Hong Kong, in China, Pu-tai, Chiayi and Cheng-his-li, Tainan City, Taiwan, south Vietnam, Gulf of Thailand and Java (Higgins & Davies, 1996; Minton & Jessop, 1999a, b, Minton et al., 2006). Long distance recoveries include birds banded in Victoria being recovered in Russia, at Yakutia, Verkhoyanskiy District, 11,812 km north of the banding site on the northern extremity of the breeding range and well to the west, on the Taimyr Peninsula, over 13,000 km from its banding location (Minton, 1996), and in China and Hong Kong (Minton, 1991).

The distribution of important sites is well known in the non-breeding period, with internationally important sites in Australia (22), Malaysia (2), Indonesia (1) and Thailand (1) (Bamford et al., 2008). In Australia, 9 sites are known to be important during migration, all in the southward period (Bamford et al., 2008). On northward migration Barter (2002) estimated that only 10% of the population use the Yellow Sea, most occurring in western Bohai Wan. However the discovery of very large numbers staging in Bohai Wan (Hong-Yan et al., 2011) suggests that the Yellow Sea is of more importance to the species than initially realised.

Threats

Threats in Australia, especially eastern and southern Australia, include ongoing human disturbance, habitat loss and degradation from pollution, changes to the water regime and invasive plants (Rogers et al., 2006; Australian Government, 2009; Garnett et al., 2011).

In the non-breeding grounds of Australia, some populations of this species occurs in highly populated areas that are vulnerable to habitat alteration. It is necessary to maintain undisturbed feeding and roosting habitat along the south-east coast and at sites on the north-west coasts used during migration for the species to survive at current population levels (Lane, 1987). Coastal development, land reclamation, construction of barrages and stabilisation of water levels can destroy feeding habitat. Pollution around settled areas may have reduced the availability of food.

Curlew sandpipers are threatened by wetland degradation in East Asia where it stages on migration (Bamford et al., 2008). Specifically this species is threatened at Bohai Bay which is being developed at a rapid rate (Murray et al., 2014). Threats at migratory staging sites include environmental pollution, reduced river flows, sea level rise, human disturbance and reclamation for tidal power plants and barrages, industrial use and urban expansion (Garnett et al., 2011; Iwamura et al., 2013).

How judged by the Committee in relation to the EPBC Act Criteria and Regulations

Criterion 1. Population size reduction (reduction in total numbers) Population reduction (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

Evidence:

Eligible under Criterion 1 A2(a) for listing as Critically Endangered.

The global population has been estimated at 1 850 000 individuals, of which about 180 000 are found in the East Asian – Australasian Flyway (Bamford et al., 2008), however, these are old data. In Australia, 115 000 individuals were thought to visit during the non-breeding period (Bamford et al., 2008), but numbers have subsequently declined (Garnett et al., 2011).

Numbers declined on Eighty-Mile Beach, WA, by c. 59% between 2000 and 2008 (Rogers et al., 2009), at the Coorong, SA, by 79% between the 1980s and 2004 (Wainwright and Christie, 2008), at sites across Queensland by 6.3% per year between 1998 and 2008 (Fuller et al., 2009), at Corner Inlet in Victoria by 3.4% per year between 1982 and 2011 (Minton et al., 2012), at Gulf St Vincent, SA, by 71% between 1981 and 2004 (Close, 2008), and by 82% across 49 Australia sites between 1983 and 2007 (BirdLife Australia *in litt.* 2011). Models suggest that this decline is due to reduced adult survival rates (Rogers and Gosbell, 2006).

Numbers in south east Tasmania have decreased by 100% in the period 1973 – 2014, with no curlew sandpipers recorded during coordinated summer counts in 2008, and 2010 – 2014 inclusive (Woehler pers. comm., 2014).

Numbers declined less severely elsewhere in the flyway. There were no clear trends in Japan between 1978 and 2008 (Amano et al., 2010), but as discussed above, Japan is not a major part of the migration route of this species.

A subsequent and more detailed assessment by a University of Queensland team (partly funded by the Department under an Australian Research Council collaborative grant), puts the species into the critically endangered category (Fuller, pers. comm., 2014). Time series data from directly observed summer counts at a large number of sites across Australia indicate a severe population decline of 75.9% over 20 years (7.5% per year; Fuller, pers. comm., 2014). This equates to a decline of 49.1% over a 10 year period, and 80.8% over 23 years, which is three generations for this species (Garnett et al., 2011).

In large part, the observed decline in curlew sandpiper numbers across Australia stems from ongoing loss of intertidal mudflat habitat at key migration staging sites in the Yellow Sea (Murray et al., 2014). As such, qualification under criterion A2 rather than A1 is warranted. However, threats are occurring locally in Australia, such as coastal development and recreational activities causing disturbance, also impact the species.

The Committee considers that the species has undergone a very severe reduction in numbers over three generation lengths (23 years for this assessment), equivalent to at least 80.8 percent and the reduction has not ceased, the cause has not ceased and is not understood. Therefore, the species has been demonstrated to have met the relevant elements of Criterion 1 to make it eligible for listing as critically endangered.

Criterion 2. Geographic distribution is precarious for either extent of occurrence AND/OR area of occupancy			
	Critically Endangered Very restricted	**Endangered Restricted**	**Vulnerable Limited**
B1. Extent of occurrence (EOO)	**< 100 km^2**	**< 5,000 km^2**	**< 20,000 km^2**
B2. Area of occupancy (AOO)	**< 10 km^2**	**< 500 km^2**	**< 2,000 km^2**
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 subpopulations; (v) number of mature individuals			
(c) Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations; (number of mature individuals			

Evidence:

Not eligible

The extent of occurrence in Australia is estimated to be 7 600 000 km^2 (stable) and area occupied 6 800 km^2 (stable; Garnett et al., 2011). Therefore, the species has not been demonstrated to have met this required element of this criterion.

Criterion 3. Small 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 subpopulation	**≤ 50**	**≤ 250**	**≤ 1,000**
	(ii) % of mature individuals in one subpopulation =	**90 – 100%**	**95 – 100%**	**100%**
(b)	Extreme fluctuations in the number of mature individuals			

Evidence:

Not eligible

The number of mature individuals in Australia is estimated to be 115 000 with a decreasing trend (Bamford et al., 2008; Garnett et al., 2011), however, these estimates are out of date and likely to be an overestimate. Therefore, the species has not been demonstrated to have met this required element of this criterion.

Criterion 4. Very small population			
	Critically Endangered Extremely low	**Endangered Very Low**	**Vulnerable Low**
Number of mature individuals	**< 50**	**< 250**	**< 1,000**

Evidence:

Not eligible

The number of mature individuals in Australia is estimated to be 115 000 with a decreasing trend (Bamford et al., 2008; Garnett et al., 2011), however, these estimates are out of date and likely to be an overestimate.

The total number of mature individuals is 115 000 which is not considered extremely low, very low or low. Therefore, the species has not been demonstrated to have met this required element of this criterion.

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**

Evidence:

Not eligible

Population viability analysis has not been undertaken

Conservation Actions

Recovery Plan

There should not be a recovery plan for this species, as approved conservation advice provides sufficient direction to implement priority actions and mitigate against key threats. Significant management and research is being undertaken at international, state and local levels.

Primary Conservation Objectives

International objectives

1. Achieve a stable or increasing population.
2. Maintain and enhance important habitat.
3. Disturbance at key roosting and feeding sites reduced.

Australian objectives

1. Achieve a stable or increasing population.

2. Maintain and enhance important habitat.
3. Disturbance at key roosting and feeding sites reduced.
4. Raise awareness of curlew sandpiper within the local community.

Conservation and Management Actions

1. Work with governments along the East Asian – Australasian Flyway to prevent destruction of key migratory staging sites.
2. Support initiatives to protect and manage key staging sites of curlew sandpiper.
3. Maintain and improve protection of roosting and feeding sites in Australia.
4. Incorporate requirements for curlew sandpiper into coastal planning and management.
5. Manage important sites to identify, control and reduce the spread of invasive species.
6. Manage disturbance at important sites when curlew sandpipers are present – e.g. discourage or prohibit vehicle access, horse riding and dogs on beaches, implement temporary beach closures.
7. Monitor the progress of recovery, including the effectiveness of management actions and the need to adapt them if necessary.

Monitoring priorities

1. Enhance existing migratory shorebird population monitoring programmes, particularly to improve coverage across northern Australia.

Information and research priorities

1. More precisely assess curlew sandpiper population size, distribution and ecological requirements particularly across northern Australia.
2. Improve knowledge about dependence of curlew sandpiper on key migratory staging sites, and wintering sites to the north of Australia.
3. Improve knowledge about threatening processes including the impacts of disturbance.

Recommendations

(i) The Committee recommends that the list referred to in section 178 of the EPBC Act be amended by **including** in the list in the Critically Endangered category:

Calidris ferruginea

(ii) The Committee recommends that there should not be a recovery plan for this species.

Threatened Species Scientific Committee

4/3/2015

References cited in the advice

Amano, T., Székely, T., Koyama, K., Amano, H., & Sutherland, W.J. (2010). A framework for monitoring the status of populations: an example from wader populations in the East Asian-Australasian flyway. *Biological Conservation* 143, 2238-2247.

Australian Government (2006). Wildlife Conservation Plan for Migratory Shorebirds. [Online]. Canberra, ACT: Department of the Environment and Heritage. Available from: http://www.environment.gov.au/biodiversity/migratory/publications/shorebird-plan.html.

Australian Government (2009). Draft significant impact guidelines for 36 migratory shorebirds. Draft EPBC Act Policy Statement 3.21. Canberra, ACT: Department of the Environment and Heritage.

Australian Government (2014). Australian Bird & Bat Banding Scheme Database, accessed 25 July 2014. Department of the Environment, Canberra.

Bamford, M., Watkins, D., Bancroft, W., Tischler, G., & Wahl, J. (2008). Migratory Shorebirds of the East Asian - Australasian Flyway: Population estimates and internationally important sites. [Online]. Canberra, ACT: Department of the Environment, Water, Heritage and the Arts, Wetlands International-Oceania. Available from: http://www.environment.gov.au/biodiversity/migratory/publications/shorebirds-east-asia.html.

Barker, R.D. & Vestjens, W.J.M. (1989). The Food of Australian Birds. 1 Non-Passerines. Lyneham, ACT: CSIRO.

Barter, M. (2002). Shorebirds of the Yellow Sea: Importance, Threats and Conservation Status. Wetlands International Global Series 9, International Wader Studies 12, Canberra, Australia

BirdLife International (2014) Species factsheet: *Calidris ferruginea*. Downloaded from http://www.birdlife.org on 23/06/2014. Recommended citation for factsheets for more than one species: BirdLife International (2014) IUCN Red List for birds. Downloaded from http://www.birdlife.org on 23/06/2014.

Christidis, L. & Boles, W.E. (2008). Systematics and Taxonomy of Australian Birds. Collingwood, Victoria: CSIRO Publishing.

Close, D.H. (2008). Changes in wader numbers in the Gulf St Vincent, South Australia, 1979-2008. *Stilt* 54, 24-27.

Dann, P. (1999a). Foraging behaviour and diets of Red-necked Stints and Curlew Sandpipers in south-eastern Australia. *Wildlife Research*. 27:61-68.

Dann, P. (1999b). Feeding periods and supratidal feeding of Red-necked Stints and Curlew Sandpipers in Western Port, Victoria. *Emu*. 99:218-222.

Delany, S. & Scott, D. (2002). Waterbird Population Estimates – Third Edition. Wetlands International Global Series No. 12, Wageningen, The Netherlands.

Fuller, R. (2014). Personal communication by email, 14 July 2014. University of Queensland.

Fuller, R.A., Wilson, H.B., Kendall, B.E., & Possingham, H.P. (2009). Monitoring shorebirds using counts by the Queensland Wader Study Group. Report to the Queensland Wader Study Group and the Department of Environment and Resource Management, Melbourne.

Garnett, S.T., Szabo, J.K., & Dutson, G. (2011). *The Action Plan for Australian Birds 2010.* Birds Australia, CSIRO Publishing, Melbourne.

Hayman, P., Marchant, J., & Prater, T. (1986). *Shorebirds. An identification guide to the waders of the world*. London & Sydney: Croom Helm.

Higgins, P.J. & Davies, S.J.J.F., (eds) (1996). *Handbook of Australian, New Zealand and Antarctic Birds. Volume Three - Snipe to Pigeons*. Melbourne, Victoria: Oxford University Press.

Hong-Yan Y, Chen B, Barter M, Piersma T, Zhou C, Li F-S & Zhang Z-W (2011). Impacts of tidal land reclamation in Bohai Bay, China: ongoing losses of critical Yellow Sea waterbird staging and wintering sites. *Bird Conservation International*, 21:241-259

Iwamura ,T., Possingham, H.P., Chadès, I., Minton, C., Murray, N.J., Rogers, D.I., Treml, E.A. & Fuller, R.A. (2013). Migratory connectivity magnifies the consequences of habitat loss from sea-level rise for shorebird populations. *Proceedings of the Royal Society B*, 281: 20130325.

Johnsgard, P.A. (1981). *The Plovers, Sandpipers and Snipes of the World*. Lincoln: Nebraska Press.

Lane, B.A. (1987). *Shorebirds in Australia*. Sydney, NSW: Reed.

Lappo, E.G., Tomkovich, P.S., Syroechkovskiy, E. (2012). *Atlas of Breeding Waders in the Russian Arctic*. Moscow: Institute of Geography, Russian Academy of Sciences.

McCarthy, E.M. (2006). *Handbook of Avian Hybrids of the World*. New York: Oxford University Press.

Minton, C. (1991). Victorian Wader Study Group Highlights. *Stilt*. 18:10.

Minton, C. (1996). Analysis of overseas movements of Red-necked Stints and Curlew Sandpipers. *Victorian Wader Study Group Bulletin*. 20:39-43.

Minton, C., & Jessop, R. (1999a). Sightings of leg-flagged waders from Victoria, Australia. Report number 7. *Stilt*. 35:43-51.

Minton, C., & Jessop, R. (1999b). Sightings of waders and terns leg-flagged in north-western Australia. Report number 6. *Stilt.* 35:52-58.

Minton, C.D.T., Jessop, R.E., Collins, P.C. and Wilson, J.R. (2006). The migratory movements of Curlew Sandpipers which visit Australia. *International Wader Studies* 19: 171-183.

Minton, C.T.D., Dann, P., Ewing, P.C., Jessop, R., Anton, P., & Clemens, R. (2012). Trends of shorebirds in Corner Inlet, Victoria, 1982-2011. *Stilt* 61:3-18.

Minton, C., & Whitelaw, J. (2000). Waders roosting on mangroves. *Stilt*. 37:23-24.

Murray, N.J., Clemens, R.S., Phinn, S.R., Possingham, H.P., & Fuller, R.A. (2014). Tracking the rapid loss of tidal wetlands in the Yellow Sea. *Frontiers in Ecology and the Environment.*

NSW Department of Environment and Climate Change (DECC) (2005). Taren Point Shorebirds - profile. [Online]. NSW DECC. Available from: http://www.threatenedspecies.environment.nsw.gov.au/tsprofile/profile.aspx?id=10800.

Rogers, D.I. 1999. What determines shorebird feeding distribution in Roebuck Bay? Capter 9 (Pp 145-179) in M. Pepping, T. Piersma, G. Pearson and M. Lavaleye (Eds). Intertidal sediments and benthic animals of roebuck Bay, Western Australia. NIOZ Report 1999-3, Texel.

Rogers, D.I. 2005. The distribution of shorebirds along Eighty-mile Beach. Chapter 10 in G.P. Pearson, T. Piersma, M. Lavaleye and R. Hickey (Eds). The long mud: Benthos and shorebirds of the foreshore of Eighty-mile Beach, Western Australia. NIOZ-Report 2005-2.

Rogers, D., Hassell, C., Oldland, J., Clemens, R., Boyle, A., Rogers, K. (2009). Monitoring Yellow Sea migrants in Australia (MYSMA): north-western Australian shorebird surveys and workshops, December 2008.

Rogers, D.I., Piersma, T., & Hassell, C.J. (2006). Roost availability may constrain shorebird distribution: exploring the energetic costs of roosting and disturbance around a tropical bay. *Biological Conservation* 133, 225-235.

Rogers, K.G., & Gosbell, K. (2006). Demographic models for Red-necked Stint and Curlew Sandpiper in Victoria. *Stilt* 50, 203-214.

Rogers, D.I., Loyn, R.H., & Greer, D. (2013). Factors influencing shorebird use of tidal flats adjacent to the Western Treatment Plant. Arthur Rylah Institute Technical Report No 250.

Tulp, I., & de Goeij, P. (1994). Evaluating wader habitat in Roebuck Bay (north-western Australia) as a springboard for northbound migration in waders, with a focus on Great Knots. *Emu*. 94:78-95.

Wainwright, P., & Christie, M. (2008). Wader surveys at the Coorong and S.E. Coastal Lakes, South Australia. *Stilt* 54, 31-47.

Woehler, E. (2014). Personal communication by email, 5 November 2014. Birds Tasmania.

Conservation Advice

Numenius madagascariensis

eastern curlew

Taxonomy

Conventionally accepted as eastern curlew *Numenius madagascariensis* Linnaeus, 1766, Scolopacidae. Other common names include Australian or sea curlew, far eastern curlew and curlew.

Monotypic, no subspecies are recognised (Bamford et al., 2008). Taxonomic uniqueness: medium (22 genera/family, 8 species/genus, 1 subspecies/species; Garnett et al., 2011).

Summary of assessment

Conservation status

Critically endangered: Criterion 1 A2,(a)

Numenius madagascariensis has been found to be eligible for listing under the following listing categories:

Criterion 1: A2 (a): Critically Endangered
Criterion 2: Not eligible
Criterion 3: Not eligible
Criterion 4: Not eligible
Criterion 5: Not eligible

The highest category for which *Numenius madagascariensis* is eligible to be listed is Critically Endangered.

Species can be listed as threatened under state and territory legislation. For information on the listing status of this species under relevant state or territory legislation, see http://www.environment.gov.au/cgi-bin/sprat/public/sprat.pl

Reason for conservation assessment by the Threatened Species Scientific Committee

This advice follows assessment of information provided by a committee nomination based on information provided in the *Action Plan for Australian Birds 2010* (Garnett et al., 2011), and experts from the University of Queensland.

Public Consultation

Notice of the proposed amendment and a consultation document were made available for public comment for 33 business days between 1 October 2014 and 14 November 2014. Any comments received that were relevant to the survival of the species were considered by the Committee as part of the assessment process.

Species Information

Description

The eastern curlew is the largest migratory shorebird in the world, with a long neck, long legs, and a very long downcurved bill. The wingspan is 110 cm and the birds weigh approximately 900 g. The head and neck are dark brown and streaked with darker brown. The chin and throat are whitish and there is a prominent white eye-ring; the iris is dark brown. The feathers of the upper parts of the body are brown, with blackish centres, and have broad pale rufous or olive-brown edges or notches. The tail is grey-brown with narrow dark banding on the feathers. The underside of the bird is dark brownish-buff, becoming paler on the rear belly. There is fine dark-brown streaking on the fore-neck and breast, which becomes thicker arrow-shaped streaks and barring on the fore-flanks. The upper belly and rear flanks have finer and sparser dark streaking. The underneath of the wing is whitish, but appears darker due to fine dark barring. The bill is dark brown with a pinkish base and the legs and feet are blue-grey.

The female is slightly larger than the male with noticeably longer bill (Higgins & Davies, 1996).

Distribution

Australian distribution

Within Australia, the eastern curlew has a primarily coastal distribution. The species is found in all states, particularly the north, east, and south-east regions including Tasmania. Eastern curlews are rarely recorded inland. They have a continuous distribution from Barrow Island and Dampier Archipelago, Western Australia, through the Kimberley and along the Northern Territory, Queensland, and NSW coasts and the islands of Torres Strait. They are patchily distributed elsewhere.

In Victoria, the main strongholds are in Corner Inlet and Western Port Bay, with smaller populations in Port Phillip Bay and scattered elsewhere along the coast. Two thirds of the birds in the Victorian population are female (Nebel et al. 2013); given that the species is monogamous, it is likely there are male-skewed non-breeding populations elsewhere, but sex-ratios have not been studied outside Victoria. Eastern curlews are found on islands in Bass Strait and along the north-west, north-east, east and south- east coasts of Tasmania. In South Australia, the species is scarce between the Victorian border and Cape Jaffa and patchily distributed from the Coorong north-west to the Streaky Bay area, and has previously been recorded in Lake Alexandrina and Lake Albert, South Australia. In southern Western Australia, eastern curlews are recorded from Eyre, and there are scattered records from Stokes Inlet to Peel Inlet. The species is a scarce visitor to Houtman Abrolhos and the adjacent mainland, and is also recorded around Shark Bay. It is also recorded on Norfolk Island and Lord Howe Island (Marchant & Higgins, 1993).

Global distribution

The eastern curlew is endemic to the East Asian – Australasian Flyway. Eastern curlews breed in Russia in southern Ussuriland, the Iman River, scattered through south, west and north Kamchatka, the lower and middle Amur River basin, the Lena River basin, between 110° E and 130° E up to 65° N, and on the Upper Yana River, at 66° N. It also breeds in Mongolia and north-eastern China

The eastern curlew is a common passage migrant in Japan, Republic of Korea, China and Indonesia, and is occasionally recorded moving through Thailand and the Malay Peninsula. During the non-breeding season a few birds occur in southern Republic of Korea, Japan and China. About 25% of the population is thought to winter in the Philippines, Indonesia and Papua New Guinea but most (estimated at 73% or 28 000 individuals) spend the non-breeding season in Australia. Eastern curlews are regular non-breeding visitors to New Zealand in small numbers, and occur rarely on Kermadec Island and the Chatham Islands (Marchant & Higgins, 1993).

Relevant Biology/Ecology

Life history

The generation time is 10.1 years (Garnett et al., 2011).

Data extracted from the Australian Bird and Bat Banding Scheme (ABBBS) reports a longevity record of 19 years, 1 month (Australian Government, 2014).

Breeding

The eastern curlew does not breed in Australia.

Eastern curlews nest in the Northern Hemisphere summer, from early May to late June, often in small colonies of two to three pairs. They nest on small mounds in swampy ground, often near where wild berries are growing. The nest is lined with dry grass and twigs. The birds may delay breeding until three to four years of age (del Hoyo et al., 1996).

General habitat

During the non-breeding season in Australia, the eastern curlew is most commonly associated with sheltered coasts, especially estuaries, bays, harbours, inlets and coastal lagoons, with large intertidal mudflats or sandflats, often with beds of seagrass (Zosteraceae). Occasionally, the species occurs on ocean beaches (often near estuaries), and coral reefs, rock platforms, or rocky islets. The birds are often recorded among saltmarsh and on mudflats fringed by mangroves, and sometimes within the mangroves. The birds are also found in coastal saltworks and sewage farms (Marchant & Higgins, 1993).

Feeding habitat

The eastern curlew mainly forages during the non-breeding season on soft sheltered intertidal sandflats or mudflats, open and without vegetation or covered with seagrass, often near mangroves, on saltflats and in saltmarsh, rockpools and among rubble on coral reefs, and on ocean beaches near the tideline. The birds are rarely seen on near-coastal lakes or in grassy areas (Marchant & Higgins, 1993).

Roosting habitat

The eastern curlew roosts during high tide periods on sandy spits, sandbars and islets, especially on beach sand near the high-water mark, and among coastal vegetation including low saltmarsh or mangroves. They occasionally roost on reef-flats, in the shallow water of lagoons and other near-coastal wetlands. Eastern curlews have occasionally been recorded roosting in trees and on the upright stakes of oyster-racks (Marchant & Higgins, 1993). At Roebuck Bay, Western Australia, birds have been recorded flying from their feeding areas on the tidal flats to roost 5 km inland on a flooded supratidal claypan (Collins et al., 2001). In some conditions, shorebirds may choose roost sites where a damp substrate lowers the local temperature. This may have important conservation implications where these sites are heavily disturbed beaches (Rogers, 1999). It may be possible to create artificial roosting sites to replace those destroyed by development (Harding et al., 1999). Eastern curlews typically roost in large flocks, separate from other shorebirds (Marchant & Higgins, 1993).

Feeding

The eastern curlew is carnivorous during the non-breeding season, mainly eating crustaceans (including crabs, shrimps and prawns), small molluscs, as well as some insects. In studies at Moreton Bay, south-east Qld, three species of intertidal decapod dominated the diet: soldier crabs (*Myctryris longicarpus*), sentinel crabs (*Macrophthalmus crassipes*) and ghost-shrimps (*Trypea australiensis*) (Zharikov and Skilleter 2004). In Victoria, ghost-shrimps are an important part of the diet (Dann 1986, 1987). In Roebuck Bay, Western Australia, the birds feed mainly on large crabs, but will also catch mantis shrimps and chase mudskippers (Rogers, 1999).

The eastern curlew is extremely wary and will take flight at the first sign of danger, long before other nearby shorebirds become nervous. The birds are both diurnal and nocturnal with feeding and roosting cycles determined by the tides. Eastern curlews find the burrows of prey by sight during the day or in bright moonlight, but also locate prey by touch. The sexual differences in bill length lead to corresponding differences in diet and behaviour (Marchant & Higgins, 1993). Eastern curlews usually feed singly or in loose flocks. Occasionally, this species is seen in large feeding flocks of hundreds (Marchant & Higgins, 1993).

Migration patterns

The eastern curlew is migratory. After breeding, they move south for the Northern Hemisphere winter. The birds migrate by day and night at varying altitudes (Marchant & Higgins, 1993).

Departure from breeding grounds

Eastern curlews leave Kamchatka Peninsula (Eastern Russia) from mid-July. There is a weak migration through Ussuriland, Russia, from mid-July to late September and birds pass through Kurile Island and Sakhalin, (Eastern Russia), from mid-July to late August (P.S. Tomkovich pers comm. in Marchant & Higgins, 1993). Fewer birds appear in continental Asia on the southern migration than on the northern migration (Dement'ev & Gladkov, 1951). Eastern curlews are commonly seen in Republic of Korea, Japan and China during August-October. Migration from the Yellow Sea to Australia is usually undertaken in a single direct flight (Minton et al., 2013). There are also records of migrants in Thailand, the Malaysian Peninsular, Singapore, the Philippines, and Borneo (Indonesia), broadly between August and December (Marchant & Higgins, 1993). The birds arrive in north-west and eastern Australia as early as July (Lane, 1987). In north-west Australia, the maximum arrival was recorded between mid-August and the end of August (Minton & Watkins, 1993). At least some birds stopover in northern Australia or Papua New Guinea before moving on to non-breeding grounds in southern Australia (Minton et al. 2013, Lane, 1987), either is a series of short flights or one long flight. Many birds arriving in eastern Australia appear to move down the coast from northern Queensland with influxes occurring on the east coast have suggested a general southward movement until mid-February (Alcorn, 1988); this is presumably dominated by late-arriving juveniles. Records from Toowoomba, Broken Hill and the Murray-Darling region in August and September suggest that some birds move overland (Marchant & Higgins, 1993) and arrival along the east and south-east Australian coasts suggests some fly directly to these areas (Alcorn, 1988). In southern Tasmania, most arrive in late August to early October; later arrivals, probably of juveniles, occur until December (Marchant & Higgins, 1993). When eastern curlews first arrive in south-eastern Tasmania they are found at a number of localities before congregating at Barilla Bay or Orielton Lagoon (BirdLife Tasmania unpubl. data).

Eastern curlews arrive in New Zealand from the second week of August until mid-November with median date mid-October (Marchant & Higgins, 1993). These relatively late arrivals suggest that the small NZ population (<20 birds) is dominated by immatures.

Non-breeding season

During the non-breeding season small numbers of eastern curlew occur in southern Republic of Korea, Japan, China and Taiwan. Unquantified numbers occur in Papua New Guinea, Borneo, and possibly Peninsular Malaysia and the Philippines (Marchant & Higgins, 1993). The majority of the eastern curlew population is found in Australia during the non-breeding season (Bamford et al., 2008), mostly at a few sites on the east and south coasts and in north-western Australia (Lane, 1987). Population numbers are stable at most sites in November or between December-February, indicating little movement during this period (Lane, 1987; Alcorn, 1988). Eastern curlews move locally between high-tide roost-sites and intertidal feeding zones (Marchant & Higgins, 1993).

Return to breeding grounds

In Australia, most eastern curlews leave between late February and March-April (Marchant & Higgins, 1993). The birds depart New Zealand from mid-March to mid-May (Marchant & Higgins, 1993). Satellite-tracking (Driscoll and Ueta 2002) and geolocation studies (Minton et al., 2013) indicate that it is usual for eastern curlew to migrate from south-eastern Australian non-breeding grounds to the northern Yellow Sea in a single flight, but that birds may take additional stops if they encounter poor migration conditions. The species has been recorded on passage in various locations mostly between March and May, arriving at Kamchatka, Russia, during May (Marchant & Higgins, 1993).

Most shorebirds including eastern curlew, spend their first and second austral (southern) winters in Australia, and some or all may also spend their third winter here before undertaking their first northward migration to the breeding grounds (Wilson, 2000). Eastern curlews probably have longer-delayed maturity than any other Australian shorebird, with many individuals not migrating north until their third year and some not migrating north until their fourth (Rogers et al. 2008).

Descriptions of migratory pathways and important sites

Internationally, the Yellow Sea is extremely important as stopover habitat for eastern curlews. It supports about 80% of the estimated flyway population on the northern migration. Counts on southwards migration appear to be lower (Barter 2002) but this probably reflects search effort and timing, given that preliminary geolocator results suggest the same staging sites in the Yellow Sea are used on both southwards and northwards migration (Minton et al., 2013). Relatively few eastern curlews pass through Japan. Thirteen sites of international importance have been identified in the Yellow Sea (six in China, six in Republic of Korea and one in North Korea). Twelve sites are known to be important during the northern migration and seven during the southern migration, with six sites (Dong Sha, Shuangtaizihekou National Nature Reserve, Ganghwa Do, Yeong Jong Do, Mangyeung Gang Hagu and Dongjin Gang Hagu) important during both (Barter, 2002).

Threats

Threats in Australia, especially eastern and southern Australia, include ongoing human disturbance, habitat loss and degradation from pollution, changes to the water regime and invasive plants (Rogers et al., 2006; Australian Government, 2009; Garnett et al., 2011).

Human disturbance can cause shorebirds to interrupt their feeding or roosting and may influence the area of otherwise suitable feeding habitat that is actually used. Disturbance to pre-migratory eastern curlews may adversely affect their capacity to migrate, as the birds will use energy reserves to avoid disturbance, rather than for migration. Eastern curlews take flight when humans approach to within 30–100 metres (Taylor & Bester, 1999), or even up to 250 metres away (Peter, 1990). Coastal development, land reclamation, construction of barrages and stabilisation of water levels can destroy feeding habitat (Close & Newman, 1984). Pollution around settled areas may reduce the availability of food (Close & Newman, 1984).

Formerly, eastern curlews were shot for food in Tasmania (Marchant & Higgins, 1993). The species has been hunted intensively on breeding grounds and at stopover points while on migration (Marchant & Higgins, 1993).

Eastern curlews are threatened by wetland degradation in the Yellow Sea where it stages on migration (Bamford et al., 2008; van de Kam et al., 2010; Murray et al., 2014). Threats along their migratory route include sea level rise, environmental pollution, reduced river flows, human disturbance and reclamation for tidal power plants and barrages, industrial use and urban expansion (Barter, 2002; Kelin and Qiang, 2006; Moores, 2006; Iwamura et al., 2013). Additional threats include disturbance at nesting sites and hunting on the breeding grounds (Barter et al., 1997).

How judged by the Committee in relation to the EPBC Act Criteria and Regulations

Criterion 1. Population size reduction (reduction in total numbers) Population reduction (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

Evidence:

Eligible under Criterion 1 A2 (a) for listing as Critically Endangered

The global population estimate was 38 000 individuals including 28 000 in Australia (Bamford et al., 2008), but numbers have recently declined (Garnett et al., 2011). This population estimate is out of date given the ongoing population declines.

Numbers appear to have declined on Eighty-mile Beach, WA by c.40% between 2000 and 2008, whereas numbers at Roebuck Bay, WA have remained relatively stable (Rogers et al., 2009). At Moreton Bay, QLD they declined by c. 2.4% per year between 1992 and 2008 (Wilson et al., 2011), across the whole of QLD they declined by c. 4.14% between 1992 and 2008 (Fuller et al., 2009), in Victoria by 2.2% per year between 1982 and 2011 (Minton et al., 2012) and in Tasmania by 80% between the 1950s and 2000 (Reid & Park, 2003) and by 40% across 49 Australian sites between 1983 and 2007 (BirdLife Australia *in litt.* 2011). An observation of over 2000 eastern curlews at Mud Islands, Port Phillip Bay in 1953 (Tarr and Launder 1954), *cf* current counts of fewer than 50 birds in Port Phillip Bay, suggests that population declines in eastern curlew may have begun well before regular shorebird counts were initiated in Australia.

An unpublished assessment of the numbers of eastern curlews at roost sites in Tasmania showed decreases of between 55% and 93%, depending on site (Woehler pers. comm., 2014). In the southeast, the decrease was 90% for the period 1964/65 – 2010/11, and in the north, the decrease was 93% between 1973/74 and 2010/11 (Woehler pers. comm., 2014). At both of these sites, and at other roost sites in Tasmania, the decreases have continued, with fewer birds seen in 2014 (Woehler pers. comm., 2014).

There are no clear trends in Japan between 1978 and 2008 (Amano et al., 2010), but this region lies outside the main migration route of eastern curlew.

A subsequent and more detailed assessment by a University of Queensland team (partly funded by the Department of the Environment under an Australian Research Council collaborative grant), puts the species into the critically endangered category (Fuller, pers. comm., 2014). Time series data from directly observed summer counts at a large number of sites across Australia indicate a severe population decline of 66.8% over 20 years (5.8% per year; Fuller, pers. comm. 2014), and 81.4 % over 30 years which for this species is equal to three generations (Garnett et al., 2011).

In large part, the observed decline in eastern curlew numbers across Australia stems from ongoing loss of intertidal mudflat habitat at key migration staging sites in the Yellow Sea (Murray et al., 2014). As such, qualification under criterion A2 rather than A1 seems warranted. However, threats are also occurring in Australia including coastal development and recreational activities causing disturbance.

The Committee considers that the species has undergone a very severe reduction in numbers over three generation lengths (30 years for this assessment), equivalent to at least 81.4 percent and the reduction has not ceased, the cause has not ceased and is not understood. Therefore, the species has been demonstrated to have met the relevant elements of Criterion 1 to make it eligible for listing as critically endangered.

Criterion 2. Geographic distribution is precarious for either extent of occurrence AND/OR area of occupancy			
	Critically Endangered Very restricted	**Endangered Restricted**	**Vulnerable Limited**
B1. Extent of occurrence (EOO)	**< 100 km^2**	**< 5,000 km^2**	**< 20,000 km^2**
B2. Area of occupancy (AOO)	**< 10 km^2**	**< 500 km^2**	**< 2,000 km^2**
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 subpopulations; (v) number of mature individuals			
(c) Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations; (number of mature individuals			

Evidence:

Not eligible

The extent of occurrence in Australia is estimated to be 30 000 km^2 (stable) and area occupied 8 500 km^2 (decreasing; Garnett et al., 2011). Therefore, the species has not been demonstrated to have met this required element of this criterion.

Criterion 3. Small 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 subpopulation	**≤ 50**	**≤ 250**	**≤ 1,000**
	(ii) % of mature individuals in one subpopulation =	**90 – 100%**	**95 – 100%**	**100%**
(b)	Extreme fluctuations in the number of mature individuals			

Evidence:

Not eligible

The number of mature individuals in Australia was estimated at 28 000 in 2008 (Bamford et al., 2008; Garnett et al., 2011), but has declined since. There are no current data available to allow assessment against this criterion. Therefore, the species has not been demonstrated to have met this required element of this criterion.

Criterion 4. Very small population			
	Critically Endangered Extremely low	**Endangered Very Low**	**Vulnerable Low**
Number of mature individuals	**< 50**	**< 250**	**< 1,000**

Evidence:

Not eligible

The total number of mature individuals was estimated at 28 000 in 2008 (Bamford et al., 2008; Garnett et al., 2011), but has declined since. The estimate is not considered extremely low, very low or low. Therefore, the species has not been demonstrated to have met this required element of this criterion.

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**

Evidence:

Not eligible

Population viability analysis has not been undertaken

Conservation Actions

Recovery Plan

There should not be a recovery plan for this species, as approved conservation advice provides sufficient direction to implement priority actions and mitigate against key threats. Significant management and research is being undertaken at international, state and local levels.

An International Single Species Action Plan will be developed and implemented across the East Asian – Australasian Flyway. Additionally, BirdLife Australia coordinates Australia's national shorebird monitoring program, Shorebirds 2020. This volunteer-based program conducts national shorebird surveys twice per year.

Primary Conservation Objectives

International objectives

1. Achieve a stable or increasing population.
2. Maintain and enhance important habitat.
3. Reduce disturbance at key roosting and feeding sites.

Australian objectives

1. Achieve a stable or increasing population.
2. Maintain and enhance important habitat.
3. Reduce disturbance at key roosting and feeding sites.
4. Raise awareness of eastern curlew within the local community.

Conservation and Management Actions

1. Work with governments along the East Asian – Australasian Flyway to prevent destruction of key migratory staging sites.
2. Develop and implement an International Single Species Action Plan for eastern curlew with all range states.
3. Support initiatives to improve habitat management at key sites.
4. Maintain and improve protection of roosting and feeding sites in Australia.
5. Incorporate requirements for eastern curlews into coastal planning and management.
6. Manage important sites to identify, control and reduce the spread of invasive species.
7. Manage disturbance at important sites when eastern curlews are present – e.g. discourage or prohibit vehicle access, horse riding and dogs on beaches, implement temporary site closures.
8. Monitor the progress of recovery, including the effectiveness of management actions and the need to adapt them if necessary.

Monitoring priorities

1. Enhance existing migratory shorebird population monitoring programmes, particularly to improve coverage across northern Australia

Information and research priorities

1. More precisely assess eastern curlew life history, population size, distribution and ecological requirements particularly across northern Australia.
2. Improve knowledge about dependence of eastern curlew on key migratory staging sites, and wintering sites to the north of Australia.

3. Improve knowledge about threatening processes including the impacts of disturbance and hunting.

Recommendations

(i) The Committee recommends that the list referred to in section 178 of the EPBC Act be amended by **including** in the list in the Critically Endangered category:

Numenius madagascariensis

(ii) The Committee recommends that there should not be a recovery plan for this species.

Threatened Species Scientific Committee

4/3/2015

References cited in the advice

Alcorn, R. (1988). Australasian Wader Study Group Regular Wader Counts Project. Interim report to June 1987: migratory waders. *Stilt* 12, 7-23.

Amano, T., Székely, T., Koyama, K., Amano, H., & Sutherland, W.J. (2010). A framework for monitoring the status of populations: an example from wader populations in the East Asian-Australasian flyway. *Biological Conservation* 143, 2238-2247.

Australian Government, (2009). Draft significant impact guidelines for 36 migratory shorebirds. Draft EPBC Act Policy Statement 3.21, Canberra, ACT: Department of the Environment and Heritage.

Australian Government, (2014). Australian Bird & Bat Banding Scheme Database, accessed 25 July 2014. Department of the Environment, Canberra.

Bamford, M., Watkins, D., Bancroft, W., Tischler, G., & Wahl, J. (2008). *Migratory Shorebirds of the East Asian - Australasian Flyway: Population estimates and internationally important sites*. [Online]. Canberra, ACT: Department of the Environment, Water, Heritage and the Arts, Wetlands International-Oceania. Available from: http://www.environment.gov.au/biodiversity/migratory/publications/shorebirds-east-asia.html.

Barter, M.A. (2002). *Shorebirds of the Yellow Sea: Importance, Threats and Conservation Status. Wetlands International Global Series No. 8, International Wader Studies 12*. Canberra, ACT: Wetlands International.

Barter, M., Fawen, Q., Sixian, T., Xiao, Y., & Tonkinson, D. (1997). Hunting of migratory waders on Chongming Dao: a declining occupation? *Stilt* 31, 19-22.

BirdLife International, (2014). Species factsheet: *Numenius madagascariensis*. Downloaded from http://www.birdlife.org on 23/06/2014. Recommended citation for factsheets for more than one species: BirdLife International (2014) IUCN Red List for birds. Downloaded from http://www.birdlife.org on 23/06/2014.

Blakers, M., S.J.J.F. Davies & Reilly, P.M. (1984). *The Atlas of Australian Birds*. Melbourne, Victoria: Melbourne University Press.

Close, D.H., & Newman, O.M.G. (1984). The decline of the Eastern Curlew in south-eastern Australia. *Emu* 84, 38-40.

Collins, P., Boyle, A., Minton, C., & Jessop, R. (2001). The importance of inland claypans for waders in Roebuck Bay, Broome, NW Australia. *Stilt* 38, 4-8.

Dann, P. (1986). The ecology of birds feeding in intertidal areas in Southern Victoria. PhD Thesis, Department of Zoology, University of Melbourne.

Dann, P. (1987). The feeding behaviour and ecology of shorebirds. Pages 10-20 in B.A. Lane, editor. *Shorebirds in Australia*. Nelson Publishers, Melbourne.

del Hoyo, J., Elliott, A., & Sargatal, J., (eds) (1996). *Handbook of the Birds of the World. Volume 3, Hoatzin to Auks*. Barcelona: Lynx Edicions.

Dement'ev, G.P., & Gladkov, N.A. (eds) (1951). *Birds of the Soviet Union, Volume 3*. Jerusalem: Israel Program for Scientific Translations.

Draffan, R.D.W., Garnett, S.T., Malone, G.J. (1983). Birds of the Torres Strait: an annotated list and biogeographic analysis. *Emu* 83, 207-234.

Driscoll, P.V. & M. Ueta. (2002). The migration route and behaviour of Eastern Curlews *Numenius madagascariensis*. *Ibis* 144, E119-130.

Fuller, R. (2014). Personal communication by email, 14 July 2014. University of Queensland.

Fuller, R.A., Wilson, H.B., Kendall, B.E., & Possingham, H.P. (2009). Monitoring shorebirds using counts by the Queensland Wader Study Group. Report to the Queensland Wader Study Group and the Department of Environment and Resource Management, Melbourne.

Garnett, S.T., Szabo, J.K., & Dutson, G. (2011). *The Action Plan for Australia Birds 2010.* Birds Australia, CSIRO Publishing, Melbourne.

Harding, J., Harding, S., & Driscoll, P. (1999). Empire Point Roost: a purpose built roost site for waders. *Stilt* 34, 46-50.

Higgins, P.J., & Davies, S.J.J.F. (eds) (1996). *Handbook of Australian, New Zealand and Antarctic Birds. Volume Three - Snipe to Pigeons*. Melbourne, Victoria: Oxford University Press.

Iwamura, T., Possingham, H.P., Chadès, I., Minton, C., Murray, N.J., Rogers, D.I., Treml, E.A. & Fuller, R.A. (2013). Migratory connectivity magnifies the consequences of habitat loss from sea-level rise for shorebird populations. *Proceedings of the Royal Society B*, 281: 20130325.

Kelin, C., & Qiang, X. (2006). Conserving migratory shorebirds in the Yellow Sea region. In Waterbirds around the World. (Eds G Boere, C Galbraith and D Stroud) p. 319. The Stationary Office, Edinburgh, UK.

Lane, B.A. (1987). *Shorebirds in Australia*. Sydney, NSW: Reed.

Marchant, S., & Higgins, P.J. (eds). (1993). *Handbook of Australian, New Zealand and Antarctic Birds. Volume 2 - Raptors to Lapwings*. Melbourne, Victoria: Oxford University Press.

Minton, C., & Watkins, D. (1993). The 1992 North-west Australia Wader Expedition. *Stilt* 22, 10-12.

Minton, C.D.T., Dann, P., Ewing, A., Jessop, R., Anton, P., & Clemens, R. (2012) Trends of shorebirds in Corner Inlet, Victoria, 1982-2011. *Stilt* 61:3-18.

Minton, C., Gosbell, K., Johns, P. Christie, M., Klaassen, M., Hassel, C., Boyle, A., Jessop, R. & Fox, J. (2013). New insights from geolocators deployed on waders in Australia. *Wader Study Group Bulletin* 120, 37-46.

Moores, N. (2006). South Korea's shorebirds: a review of abundance, distribution, threats and conservation status. *Stilt* 50, 62-72.

Murray, N.J., Clemens, R.S., Phinn, S.R., Possingham, H.P., & Fuller, R.A. (2014). Tracking the rapid loss of tidal wetlands in the Yellow Sea. *Frontiers in Ecology and the Environment.*

Nebel, S., Rogers, K.G., Minton, C.D.T., and Rogers, D.I. (2013). Is geographic variation in the size of Australian shorebirds consistent with hypotheses on differential migration? *Emu* 13, 99-111.

Peter, J.M. (1990). Bird Study in the Nooramunga: The Possible Effects of Oyster Farming. *RAOU Report Series* 74, 1-18.

Reid, T., & Park, P. (2003). Continuing decline of Eastern Curlew, *Numenius madagascariensis*, in Tasmania. *Emu* 103, 279-283.

Rogers, D. (1999). Roost choice in the waders of Roebuck Bay: is avoiding heat stress their main consideration? *Stilt* 35, 65.

Rogers, D., Hassell, C., Oldland, J., Clemens, R., Boyle, A., & Rogers, K. (2009). Monitoring Yellow Sea migrants in Australia (MYSMA): north-western Australian shorebird surveys and workshops, December 2008.

Rogers, D.I, Piersma, T., & Hassell, C.J. (2006). Roost availability may constrain shorebird distribution: exploring the energetic costs of roosting and disturbance around a tropical bay. *Biological Conservation* 133, 225-235.

Rogers, D.I., Minton, C.D.T., Boyle, A.N., Hassell, C.J. & Silcocks, A. (2008). Growing up slowly by the sea-side: Age of first northwards migration of shorebirds from Australian non-breeding grounds'. In D.I., Rogers, *Hidden costs: challenges faced by migratory shorebirds living on intertidal flats*. PhD Thesis, Charles Sturt University.

Tarr, H., & Launder, J. (1954). Bird Observer.

Taylor, I.R., & Bester, A. (1999). The response of foraging waders to human recreation disturbance at Rhyll, Phillip Island, Victoria. *Stilt* 35, 67.

Thomas, D.G. (1968). Waders of Hobart. *Emu* 68, 95-125.

Van de Kam, J., Battley, P.F., McCaffery, B.I., Roger, D.I., Hong, J-S., Moores, N., Ki, J.-Y., Lewis, J., & Piersma, T. (2010). Invisible Connections: Why Migrating Shorebirds Need the Yellow Sea. CSIRO Publishing, Melbourne.

Wilson, J.R. (2000). The northward movement of immature Eastern Curlews in the austral winter as demonstrated by the population monitoring project. *Stilt* 36, 16-19.

Woehler, E. (2014). Personal communication by email, 5 November 2014. Birds Tasmania.

The Minister approved this conservation advice and included this species in the Vulnerable category, effective from 5 May 2016

Conservation Advice

Macroderma gigas

ghost bat

Note: The information contained in this Conservation Advice was primarily sourced from 'The Action Plan for Australian Mammals 2012' (Woinarski et al., 2014). Any substantive additions obtained during the consultation on the draft have been cited within the advice. Readers may note that Conservation Advices resulting from the Action Plan for Australian Mammals show minor differences in formatting relative to other Conservation Advices. These reflect the desire to efficiently prepare a large number of advices by adopting the presentation approach of the Action Plan for Australian Mammals, and do not reflect any difference in the evidence used to develop the recommendation.

Taxonomy

Conventionally accepted as *Macroderma gigas* (Dobson 1880).

Macroderma is a monotypic genus endemic to Australia. There is a possibility that *Macroderma* exists in Papua New Guinea (Filewood 1983), but this has never been confirmed. The ghost bat is the largest species in the family and comprises several disjunct subpopulations across northern Australia.

A second subspecies from the Kimberley, *M. gigas saturata*, was described by Douglas (1962) using diagnoses based on pelage and skin colour. However, it has now been synonymised with *M. gigas* (Koopman 1984; Simmons 2005). Studies of morphological and genetic variation across the species' distribution found clinal variation in size (northern ghost bats were smaller; Hand & York 1990), and a high degree of population subdivision with greater connectedness amongst colonies in northern subpopulations (Worthington Wilmer et al., 1994, 1999). However, these findings were not suggested as a basis for subspecific taxonomic distinctness, and no subspecies are recognised.

Summary of assessment

Conservation status

Vulnerable: Criterion 1 A2(b)(c)(d), A3(b)(c)(d), A4(b)(c)(d) and Criterion 3 C1

Species can be listed as threatened under state and territory legislation. For information on the listing status of this species under relevant state or territory legislation, see http://www.environment.gov.au/cgi-bin/sprat/public/sprat.pl.

Reason for conservation assessment by the Threatened Species Scientific Committee

This advice follows assessment of new information provided to the Committee to list *Macroderma gigas.*

Public Consultation

Notice of the proposed amendment and a consultation document was made available for public comment for 40 business days between 30 September 2015 and 25 November 2015. Any comments received that were relevant to the survival of the species were considered by the Committee as part of the assessment process.

Species Information

Description

The ghost bat is the largest microchiropteran bat in Australia, with a head and body length of 10−13 cm and a forearm length of 10−11 cm. It is Australia's only carnivorous bat. Its fur is light to dark grey above and paler below. It has long ears which are joined together, large eyes, a simple noseleaf and no tail (Richards et al., 2008).

Distribution

Fossil data show that the ghost bat was once distributed widely over much of Australia except Victoria and Tasmania, including the arid zone, but contracted northwards during the Holocene period (Molnar et al., 1984; Churchill & Helman 1990). A study that combined information from ancient DNA obtained from remains in extinct southern populations, newly-generated and existing genetic data from extant northern populations, and ecological niche modelling based on past and present climatic conditions (Thomson et al., 2012), suggested that the ghost bat expanded southwards during periods of higher humidity (interglacials) and contracted northwards in response to increasing aridity (e.g. preceding the last glacial maximum). The combined analyses support previous statements that the ghost bat is a geographically relictual species in southern, arid landscapes, present only because caves provide suitable roost microclimates.

At the time of European settlement, arid zone subpopulations remained. Since the arrival of Europeans, ghost bats have contracted further northwards, with much of their arid zone distribution disappearing in the past few decades (Molnar et al., 1984; Churchill & Helman 1990). Burbidge et al. (1988) reported that western desert Aboriginal people stated that ghost bats only ever occurred in a few favourable areas and that they were still present. However, searches of several central Australian sites where they once occurred have since failed to locate any (Churchill & Helman 1990). The last arid zone specimen was collected in 1961 (Butler 1962). The major range contraction from central Australia happened more than three generations (24 years) ago.

The species' current range is discontinuous, with geographically disjunct colonies occurring in the Pilbara (Armstrong & Anstee 2000; McKenzie & Bullen 2009), Kimberley (including several islands; McKenzie & Bullen 2012), northern Northern Territory (including Groote Eylandt), the Gulf of Carpentaria (Australian Wildlife Conservancy 2010), coastal and near coastal eastern Queensland from Cape York to near Rockhampton (Richards et al., 2008), and western Queensland (including Riversleigh and Cammoweal districts; Bullen pers. comm., 2015). Burbidge et al. (2009), using modern, historical and subfossil data, found that the ghost bat occurred in 37 of Australia's 85 bioregions, and that it was extinct in 12. Only 14 breeding sites are currently known (Worthington Wilmer 2012).

Populations are highly structured, being genetically distinct at both regional and local scales (Worthington Wilmer et al., 1994, 1999; Armstrong et al., in prep). Populations at the southern limits of the species' range are geographically isolated and separated by a minimum distance of 300 km. This geographic isolation is reflected in the genetic data with populations at Mt Etna, Cape Hillsborough, and Camooweal in Queensland, and the Pilbara in Western Australia, being highly divergent genetically, and implies virtually no movement of individuals between these sites (Worthington Wilmer et al., 1999). Populations within the Northern Territory and far north Queensland are also highly distinct from each other and other population centres (Worthington Wilmer et al., 1999), while the Kimberley bats are distinct from all other Australian populations with genetic structure evident in the Kimberley populations (Worthington Wilmer 1996).

Population genetic studies indicate a high degree of female philopatry (remaining in, or returning to, an individual's birthplace) at natal roosts based on mitochondrial DNA markers; gene flow within regions mediated by male movements was also suggested from nuclear microsatellite markers (Worthington Wilmer et al., 1994, 1999). Northern groups had higher heterozygosity and less marked phylogeographic structure than southern groups, which was interpreted to be a consequence of the limited availability and greater separation of roost sites with suitable microclimates in more arid areas. Recent studies that have built on the work by Worthington Wilmer et al. (1994, 1999), by adding individuals from the Pilbara and Kimberley regions, have also highlighted the distinctness of these two subpopulations, high female philopatry, and gene flow within regions arising from male movements (K. Armstrong et al., pers. comm., cited in Woinarski et al., 2014). Losses of sites containing breeding females have the potential to reduce the area of occupancy and population size significantly.

Relevant Biology/Ecology

Ghost bats are the largest microchiropteran bat in Australia and the second largest in the world, weighing up to 150 g and having a wingspan of 60 cm. They currently occupy habitats ranging from the arid Pilbara to tropical savanna woodlands and rainforests. During the daytime they roost in caves, rock crevices and old mines. Roost sites used permanently are generally deep natural caves or disused mines with a relatively stable temperature of 23°−28°C and a moderate to high relative humidity of 50−100 percent (Pettigrew et al., 1986; Churchill & Helman 1990; Churchill 1991; Armstrong & Anstee 2000; J. Toop unpublished data). They are carnivores, with a broad diet comprising small mammals including other bats, birds, reptiles, frogs and large insects (Pettigrew et al., 1986; Schulz 1986; Boles 1999; J. Toop unpublished data). The proportion of food items in the diet varies with availability. At Pine Creek in the Northern Territory, diet predominantly comprised birds as large as the dollarbird (*Eurystomus orientalis*), which weighs 125−140 g (Schulz, 1986; Pettigrew et al., 1986). At Mount Etna, diet has at times been mostly large insects, while at other times the prey included vertebrates such as birds, bats, rats and mice (J. Toop, unpublished data).

The ghost bat has a surface foraging strategy with two modes. It perches in vegetation to ambush passing prey (either on the ground or in the air), and it also gleans surfaces such as the ground while in flight. Its echolocation calls show wide variation (McKenzie & Bullen 2009). Tidemann et al. (1985) found that foraging areas were centred, on average, 1.9 km from the daytime roost. The mean size of foraging areas was 61 ha and tagged bats generally returned to the same areas each night. Hunting behaviour within foraging areas consisted of observation at vantage points with brief sallies to capture prey (mostly insects on the ground), though hawking of flying insects was also observed. Vantage points were changed about every 15 minutes during foraging periods, and the mean distance between them was 360 m. Foraging areas were not exclusive; there was overlap between the ranges of several tagged individuals, and in one case an area was used by 20 bats.

Hoyle et al. (2001), who studied the southern-most known colony in Queensland, found that female bats gave birth to a single young in late spring, but only 40 percent (22–70%, 95% confidence interval (CI)) of females bred in their second year, increasing to 93 percent (87–97%, 95% CI) for females ≥ 2 years old. Sixty-five percent of juveniles captured were female. Annual adult survival ranged 0.57–0.77 for females and 0.43–0.66 for males, and was lowest over winter–spring and greatest in autumn–winter. Juvenile survival for the first year ranged 0.35–0.46 for females and 0.29–0.42 for males. Adult survival varied among seasons, and was negatively associated with rainfall but not associated with temperature apart from being lower in late winter. Poor survival may result from the inferior daytime roosts that bats must use if water seepage forces them to leave their normal roosts. Although these age-specific rates of fecundity and survival suggested a declining population, mark-recapture estimates of the population trend indicated stability over the study period. Counts at daytime roosts also suggested a population decline, but were considered unreliable because of an increasing tendency of bats to avoid detection. At Mount Etna, Toop (1985) found that pregnant females congregated in the warmest caves and gave birth over a month commencing in mid-October. As caves became warmer as summer progressed, some mothers shifted the young to other caves. Juvenile bats commenced flying at seven weeks with all young capable of flight by the end of January.

Ghost bats move between a number of caves seasonally or as dictated by weather conditions, and require a range of cave sites (Hutson et al., 2001). Most breeding sites appear to require multiple entranced caves (L. Hall pers. comm., cited in McKenzie & Hall 2008). Ghost bats disperse widely when not breeding, but concentrate in a relatively few roost sites when breeding. Few of these sites are known (Richards et al., 2008; Worthington Wilmer 2012), and most are not protected or managed.

Roost sites include caves, rock crevices and disused mine adits. In the Hamersley Range in the Pilbara, preferred roosting habitat appears to be caves beneath bluffs of low rounded hills composed of Marra Mamba geology, and larger hills of Brockman Iron Formation; in the eastern Pilbara, caves beneath bluffs composed of Gorge Creek Group geology and granite rockpiles are preferred (Armstrong & Anstee 2000). The species' persistence in the arid Pilbara depends on the physiologically benign day roosts found deep underground in humid, temperature-stable caves (Leitner & Nelson 1967; Hall et al., 1997; Armstrong & Anstee 2000; McKenzie & Bullen 2009).

Ghost bats are easily disturbed when roosting. Young may be dislodged by adults in rapid take-offs (J. Toop, unpublished data) and may not return to the roost site (K. Armstrong pers. comm., cited in Woinarski et al., 2014). This makes counting individuals at roost sites difficult and repeated counts may be unreliable (Armstrong 2010). Such susceptibility to disturbance also threatens the viability of roosts with unregulated human visitation, including surveys which target caves and may inadvertently flush individuals into daylight.

Females breed at an age of two to three years (Hoyle et al., 2001). Longevity in the wild is unknown, but is likely to be somewhat less than the maximum 22.6 years in captivity (AnAge 2012). Generation time is assumed to be 8 years (Woinarski et al., 2014).

Threats

The key threat to the ghost bat is habitat loss and degradation due to mining activities (McKenzie & Hall 2008; Qld DEHP 2015). The species' slow reproductive rate, and the lack of suitable habitat which restricts its movement, renders it vulnerable to threats and localised extinctions (Qld DEHP 2015). The genetic isolation of each subpopulation suggests areas are unlikely to be recolonised if a local extinction occurs (Qld DEHP 2015).

Threats to the ghost bat are outlined in the table below (Woinarski et al., 2014).

Threat factor	Consequence rating	Extent over which threat may operate	Evidence base
Habitat loss (destruction of, or disturbance to, roost sites and nearby areas) due to mining	Severe	Moderate	Mt Etna and the surrounding area contain breeding sites, some of which have been destroyed; declines were reported at Mt Etna following mining; Mt Etna is now protected in a national park and visited by tourists (Worthington Wilmer 2012). Mount Consider cave west of Cairns has been destroyed; other sites are still vulnerable; limestone mining is a threat in Cooktown. Many Pilbara roosts are vulnerable to iron ore mining and the deterioration and disturbance of old underground gold and copper mines.

Disturbance of (human visitation at) breeding sites	Moderate-severe	Moderate	Ghost bats are easily disturbed and may abandon sites where disturbance occurs (K. Armstrong pers. comm., cited in Woinarski et al., 2014). Minor disturbances by approaching vehicles and people may result in bats moving to alternative roost sites (Bullen pers. comm., 2015). Larger disturbances by recreational cavers or ecologists entering caves may cause the loss of pups and/or abandonment of roost sites (Bullen pers. comm., 2015).
Modification to foraging habitat	Moderate	Moderate	Vegetation simplification can impact on foraging strategies and productive riparian sites. Foraging bats search for prey from vantage points in trees before making short flights to capture prey (Tidemann et al.,1985). To persist in an area, small colonies require a group of caves/shelters that provide alternative day and night roost sites, and a gully or gorge system that opens onto a plain or riparian line that provides good foraging opportunities, typically less than 5 km from the diurnal roost site (Bullen pers. comm., 2015). Livestock grazing, fire and weed encroachment can degrade habitat (Qld DEHP 2015); some population declines could be attributable to prey lost through habitat modification by fire and livestock (Duncan et al., 1999).
Collision with fences, especially those with barbed wire	Moderate	Moderate	Ghost bats have low fecundity and survival (Hoyle et al., 2001). They often fly at about fence height and substantial numbers are known to be killed when colliding with fencing wire (Armstrong & Anstee 2000; McKenzie & Bullen 2009). A single fence near a colony can effectively remove all of these individuals given enough time, and has been observed in the Pilbara (Armstrong & Anstee 2000; Armstrong pers. comm., 2015).
Collapse or reworking of old mine adits	Minor-moderate	Minor-moderate	Many of the known nursery roosts are in old mine workings that are collapsing, flooding or subject to disturbance (Hall et al., 1997; Armstrong 2001); e.g. the Pine Creek colony roosts in an adit that is in danger of collapse (Richards et al., 2008).
Contamination by mining residue at roost sites	Moderate	Moderate	Several roosting sites in old mines have high levels of pollutants that may reduce rates of survival or reproduction.

Disease	Unknown	Unknown	A possible herpes-type virus appears to be affecting the Mt Etna population, but the pathology is yet to be confirmed (J. Augusteyn pers. comm., cited in Woinarski et al., 2014).
Poisoning by cane toads	Severe	Moderate (may become Moderate-Entire)	There is evidence of ghost bats preying upon cane toads in Kakadu NP; bats have been found dead with chewed toads in their throats (White & Bullen pers. comm., cited in Qld DEHP 2015). There has been a significant reduction in numbers of ghost bats in the Riversleigh district, western Queensland, apparently due to the consumption of cane toads (Bullen pers. comm., 2015). Genetic work indicates that the ghost bat is unable to tolerate bufotoxins (Shine et al., in review, cited in Armstrong pers. comm., 2015).
Competition for prey with foxes and feral cats	Unknown	Unknown	Some population declines could be attributable to competition for prey with foxes and feral cats (Duncan et al., 1999).

How judged by the Committee in relation to the EPBC Act Criteria and Regulations

Criterion 1. Population size reduction (reduction in total numbers)
Population reduction (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

Evidence:

Eligible under Criterion 1 A2(b)(c)(d), A3(b)(c)(d), A4(b)(c)(d) for listing as Vulnerable

Woinarski et al. (2014) estimate the population size of the ghost bat to be fewer than 10 000 mature individuals, with an estimated continuing decline of greater than 10 percent in 24 years (three generations). There is evidence of significant declines in some parts of the species' distribution.

Western Australia

Ghost bats occur in the Pilbara and the Kimberley, with abandoned mine adits (horizontal tunnels) comprising a significant proportion of the known roost sites (Woinarski et al., 2014). The presence of mines may have allowed the species to extend its range and expand its population size in the past (e.g. Worthington Wilmer et al., 1999). However, many disused mines are now collapsing or being open cut and reworked (Armstrong 2001, 2011; WA DPaW 2015).

There is a possibility of population decline following the loss of some root sites in the Pilbara (Armstrong pers. comm., 2015). Most of the population in the Pilbara region is known from six historical mine workings: Bamboo Creek, Bulletin, Comet, Klondyke Queen, Lalla Rookh and All Nations mines (Armstrong pers comm., 2015). In the past these populations probably had over 1000 individuals (Armstrong & Anstee 2000). Two of them (All Nations and Bulletin mine) appear to have now disappeared; the remaining four mines show evidence of collapse, flooding and human intrusion and are part of active mineral exploration leases, and may have decreased in size (Armstrong pers comm., 2015). The other smaller colonies are found in caves and relatively small adits, with colony sizes typically less than 10 (Armstrong and Anstee 2000; Armstrong pers. comm., 2015).

In the Pilbara, most known breeding sites of the ghost bat are confined to underground gold/copper mines that are now collapsing or being open cut, and to caves in banded ironstone strata that may be mined out over the next 30−50 years. On current trends, most of its Pilbara roost sites may be destroyed over the next 30 years (Woinarski et al., 2014). Numbers are likely to decline by over 30 percent in Western Australia in the future with local extinction in areas such as the central and eastern Hamersley Range, with the extent of occupancy likely to decline by over 10 000 km^2 (Bullen pers. comm., 2015). However, barbed wire fences are being replaced in crucial areas and breeding sites are being identified for protection (WA DPaW 2015), which may reduce the current rate of decline.

The Kimberley colonies (containing approximately two-thirds of the state's ghost bat population) are likely to be relatively stable, as little mining or habitat destruction occurs in the region, with cane toads the main threat. However, limited surveys have been undertaken in the Kimberley (WA DPaW 2015), and it is unclear to what extent cane toads will affect these populations in the future if cane toads advance further into the Kimberley.

Northern Territory

Populations in Kakadu National Park are believed to have declined by more than 90 percent since the arrival of cane toads in 2001. No formal surveys in Kakadu National Park were undertaken prior to 2014, but informal surveys and approximate counts were undertaken by rangers, with the most reliable undertaken in the 1980s (Table 1). Surveys undertaken in 2014‑2015 show that many of the largest roosting areas are now abandoned, including the largest colonial site at Ngarradji Warde Djobkeng (Table 1; White et al., in prep). The remaining colonies are reduced and in areas remote from waterholes (A. White unpublished data, cited in Qld DEHP 2015).

Table 1. Population estimates for major ghost bat sites in Kakadu National Park. Other sites not listed are small, day roosting sites. (A. White pers. comm., 2016.)

Location	1984-1986 estimates	2014-2015 surveys	Known breeding site
Ngarraddj Warde Djobknong	800+	0	Yes
Nawurlandja	30-50	1	Yes
Rockholes Mine	30-50	0	No
Blue Rocks Caves (Caves 1-6)	50-100	18	Yes
Hawk Dreaming (Caves 1-3)	50+	22	Yes
Jabiru Dreaming	30	0	No
Riflefish Dreaming	20	0	No

Counts have been undertaken at Pungalina, now owned and managed by the Australian Wildlife Conservancy, from 2005 to 2012. The population appeared to be stable throughout this period. A few ghost bat carcasses were found in 2012; it is unclear whether these can be attributable to cane toad poisoning as cane toads arrived in Pungalina several years before 2005 (N. White pers. comm., 2015b).

Milne & Pavey (2011) considered the species to be relatively common and secure in the wet dry tropics of the Northern Territory. However, the largest known breeding site at Kohinoor Adit in Pine Creek (Pettigrew et al., 1986) faces threats from unregulated human visitation, potential mine collapse and possibly contaminated water (Woinarski et al., 2014) and may be in decline (WA DPaW 2015; Qld DEHP 2015). Grant et al. (2010) summarised the counts at Kohinoor Adit (Table 2). A count was also undertaken in 2013 using a thermal video camera and missile tracking software (Armstrong pers. comm., 2015). Sampling precision has varied with methods used, and counts vary depending upon the season of count and breeding stage (Woinarski et al., 2014). However, the counts suggest that numbers may have declined by more than 30 percent over the past 24 years (three generations).

Table 2. Counts of ghost bats at various dates at the largest known breeding site, Kohinoor Adit.

Date	Count
July 1981	300
May 1983	445
June 1984	780
May 1985	1100
April 1987	1300
February 1988	1400
August 1988	1300
January 1990	1500
July 2010	564
December 2013	550

Queensland

The Queensland subpopulations are located in 4−5 highly disjunct localities. Data are available for four of the five main colonies, and all are in decline (Table 3) (Qld DEHP 2015). No information is available for the Mitchell Palmer colony. Limited information is available for the remaining colonies, but most are considered to be small with fewer than 50 individuals; it is possible the entire Queensland population is in decline but further information is required to confirm this (Qld DEHP 2015). The Boodjamulla (Lawn Hill and Riversleigh) population is now thought to be extinct (A. White pers. comm., cited in Qld DEHP 2015).

Table 3. Data for 4 Queensland subpopulations, showing decline (Qld DEHP 2015, with additions).

Subpopulation	Previous estimate	Recent estimate
Mt Etna	170 (2011/12 estimate; Worthington Wilmer 2012)	40 (number of bats seen in 2013; Augusteyn et al., in prep)
Cape Hillsborough	180 (2011/12 estimate; Worthington Wilmer 2012)	50 (inferred from multiple cave visits 2011-2014; Cali pers. comm., cited in Qld DEHP 2015)
Camooweal	160−180 (2013 estimate; Qld DEHP 2015)	50−100 (Armstrong & White pers. comm., cited in Qld DEHP 2015)
Kings Plains	167 (1995 direct count estimate by Les Hall; Hughes pers. comm., 2015)	108 (2014 direct count estimate by Peter Bannink; Hughes pers. comm., 2015)

At Mount Etna only 26 individuals were captured over several months, whereas Worthington Wilmer (1996) caught 25 individual bats over two nights in 1993 at a similar time of year, at the same site and using the same methodology (Woinarski et al., 2014). Preliminary results from a genetic coalescence study suggested an effective population size of 15−30 depending on the method used (J. Augusteyn pers. comm., cited in Woinarski et al., 2014). The average age of the Mt Etna colony is around five years, with each pair of successful breeding individuals only just achieving population replacement (Toop & Davies, unpublished). Recent trapping of the Cape Hillsborough wintering roost also indicates that the wintering population is declining when compared with numbers caught and recorded from these caves from the mid 1970s to early 1990s (M. Cali pers. comm., cited in Woinarski et al., 2014). The Mt Etna population, and probably the Cape Hillsborough population also, is genetically isolated and too small to survive as a viable population, and will likely become extinct (N. White pers. comm., 2015a).

Conclusions

A summary of past and projected declines over the past 24 years (1992-2016), based on the data provided above, are summarised in Table 4.

Table 4. Summary of above data

Population	**Past population size**	**Current population size**	**Past decline**	**Decline over a 24 year period (may include past and present)**
Pilbara, WA	Likely >2000 based on current population estimate and past decline	1300-2000	Likely >30% as 2 out of 6 sites have disappeared, with decline in the others	>>30% (inferred from threats)
Kimberley, WA	3000-4000	3000-4000	0% (inferred)	>10% (inferred from future threats and the impacts of cane toads in Kakadu)
Kakadu, NT (subset: 7 populations)	1010-1100	41	96-96%	90% (ongoing threats)
Kohinoor Adit, NT	1500	550	63%	60% (ongoing threats)
Queensland (subset: 4 populations)	677-697	248-298	56-64%	60% (ongoing threats)
TOTAL	8187-9297	5139-6889	16-45%	>30% (ongoing threats)

The Committee considers that the species has undergone a substantial reduction in numbers over three generation lengths (24 years for this assessment), equivalent to at least 30 percent and the reduction has not ceased, and the cause has not ceased. Therefore, the species 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 km^2**	**< 5,000 km^2**	**< 20,000 km^2**
B2. Area of occupancy (AOO)	**< 10 km^2**	**< 500 km^2**	**< 2,000 km^2**
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 subpopulations; (v) number of mature individuals			
(c) Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations; (iv) number of mature individuals			

Evidence:

Not eligible

The extent of occurrence is estimated at 3 989 300 km^2, and the area of occupancy estimated at 1104 km^2. These figures are based on the mapping of point records from 1996 to 2016, obtained from state governments, museums, CSIRO and the Australian Wildlife Conservancy. 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 (DotE 2016). Mapped point records from 1966 to 1996, which give an EOO of 5 649 306 km^2 and an AOO of 1952 km^2 (DotE 2016), show that the historical distribution was much larger.

The EOO is currently stable in the Pilbara but continues to decline behind the cane toad front in the Kimberley, Northern Territory and Queensland (Bullen pers. comm., 2015). The area of occupancy is continuing to decline (Woinarski et al., 2014). However, the ghost bat occurs at more than 10 locations and does not suffer extreme fluctuations (Woinarski et al., 2014). Populations are fragmented, but not considered severely fragmented (other than in Queensland) as there is likely to be interchange among colonies within, although not between, other parts of the range (McKenzie & Hall 2008).

Following assessment of the data the Committee has determined that the geographic distribution is very restricted, and there is a continuing decline in the population and distribution. However, the distribution is not severely fragmented and there is no evidence of extreme fluctuations. Therefore, the species has not met the required elements of this criterion.

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 subpopulation	≤ 50	≤ 250	≤ 1,000
(a) (ii) % of mature individuals in one subpopulation =	90 – 100%	95 – 100%	100%
(b) Extreme fluctuations in the number of mature individuals			

Evidence:

Eligible under Criterion 3 C1 for listing as Vulnerable

Although the ghost bat can be counted readily when it leaves caves and mine roosts after dusk because of its large size and pale colour, there are no robust measures of abundance across its full range. Monitoring of colony size has been conducted mostly on an ad-hoc basis over the past three decades at certain large colonies, and data have been collected from some colonies over several years (Woinarski et al., 2014).

McKenzie and Hall (2008) estimated the total population size to be 7000−9000 individuals, with differences amongst the regional subpopulations. Worthington Wilmer (2012) stated that, based on known colonies and without projections for unknown colonies, counts for Australia ranged from 4000 to 6000 individuals (750−850 in Queensland, 2500−3500 in the Northern Territory and about 1500 in Western Australia). Available population data are presented below.

<u>Western Australia</u>

Hall et al. (1997) reported the following subpopulation size data from mines in the Pilbara:

- Comet: 35 (26 April 1981); 37+ (14 October 1993); 100+ (19 July 1996)
- Klondyke: 40 (1 May 1981); 98+ (24 April 1994); 20+ (14 July 1994); 40+ (18 July 1995); counts by Armstrong (2010) varied between 107 and 366 for the period 12 June 2011 to 5 July 2001
- Bulletin: 406 (23 April 1994); 200+ (18 July 1995).

Armstrong and Anstee (2000) estimated 1200 individuals to occur in the Pilbara. However, surveys for environmental impact assessments have discovered several larger colonies in the past decade (Armstrong 2011) and activities associated with mining have had an undocumented effect at several known roost sites (K. Armstrong pers. comm., cited in Woinarski et al., 2014). McKenzie and Bullen (2009) commented on the apparent commonness of the ghost bat after recording ghost bats at 21 of their 24 survey areas in the Pilbara, and in all four Pilbara sub-regions, though diurnal roosting and colony sizes were not examined explicitly and their acoustic detection method was not optimal for this species.

Surveys since 2009 indicate that the Pilbara populations exist in two regions: the Chichester subregion with a population of approximately 1500, and the Hamersley subregion with a population of approximately 350 (Bullen pers. comm., 2015). In the Chichester subregion (eastern Pilbara), ghost bats occur mostly in medium to large groups in historical underground mines, most of which appear to be breeding sites; ghost bats are spread across the Hamersley Range in a large number of small groups of less than 20 (Armstrong & Anstee 2000; Bullen pers. comm., 2015). The current population size in the Pilbara is estimated to be 1300−1900 individuals (Armstrong pers. comm., 2015) or 1500−2000 individuals (Bullen pers. comm., 2015).

In the Kimberley a population size of around 3000−4000 individuals has been inferred (McKenzie & Hall, 2008). The species has been recorded on six Kimberley Islands which, at the date of this assessment, were last visited in February 2010 (McKenzie & Bullen 2012).

The total population size in Western Australia (comprising the Pilbara and Kimberley) is therefore estimated at 4300−6000 individuals.

Queensland

In Queensland the population size has been estimated at fewer than 1000 individuals (Woinarski et al., 2014), and possibly as low as 470−680 individuals excluding the Calvert River / Pungalina population on the Northern Territory/Queensland border (Table 5) (Qld DEHP 2015).

Table 5. Population estimates for Queensland (Qld DEHP 2015).

Subpopulation	**Most recent population estimate**
Mt Etna	40
Cape Hillsborough	50
Camooweal	50-100
Kings Plains (Cooktown)	108
Mt Isa/Cloncurry	50
Mitchell Palmer	50
Cape Melville/ McIlwraith	20
Blackbraes/Chudleigh	50
Wet Tropics	50

On Cape York Peninsula, breeding sites are known at Mitchell-Palmer limestone and Kings Plains station, with a suspected site near the Iron Range (Reardon et al*.,* 2010). Other available Queensland population estimates are of 150 at Girringun-Gugu Badhun West of Ingham / Cardwell and 500 at Kuku Nyungkul – Kuku Bubogun south of Cooktown (C. Clague pers. comm., cited in Woinarski et al., 2014).

Northern Territory

The total population in the Northern Territory is estimated to be 2500−3500 individuals, based on counts at known colonies (Worthington Wilmer 2012). The population in Pungalina, just over the border from Queensland, is estimated to be 100 from counts undertaken from 2005 to 2012 (N. White pers. comm., 2015b). The population at Kohinoor Adit is estimated to be 550 (Armstrong pers. comm., 2015), and at Kakadu around 100 (A. White pers. comm., 2016).

Conclusions

Woinarski et al. (2014) estimate the total population size to be fewer than 10 000 individuals, based on a combination of counts of colony size at some roost sites plus calculations based on area of occupancy. There is a projected continuing decline of greater than 10 percent in a future 24 year (three generation) period (Woinarski et al., 2014; also see Criterion 1). It is unknown whether the number of mature individuals in each subpopulation is less than 1000, as colony sizes in the Kimberley are unknown.

The Committee considers that the estimated total number of mature individuals of this species is limited, and the population is likely to decline at a substantial rate of 10 percent in the next three generations due to a decline in extent of occurrence, area of occupancy, habitat and number of locations. Therefore, the species has met the relevant elements of Criterion 3 to make it eligible for listing as Vulnerable.

Criterion 4. Number of mature individuals			
	Critically Endangered Extremely low	**Endangered Very Low**	**Vulnerable Low**
Number of mature individuals	**< 50**	**< 250**	**< 1,000**

Evidence:

Not eligible

The population size is estimated at 7000−9000 mature individuals (McKenzie & Hall 2008); see information provided under Criterion 3.

The total number of mature individuals is not considered extremely low, very low or low. Therefore, the species has not met this required element of this criterion.

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**

Evidence:

Not eligible

Population viability analysis has not been undertaken.

Conservation Actions

Recovery Plan

The Committee recommends that there should be a recovery plan for the ghost bat. Stopping decline and supporting recovery of the species is complex, due to the requirement for a high level of planning to abate the threats, a high level of support by key stakeholders, and a high level of prioritisation. Existing mechanisms are not adequate to address these needs.

Primary Conservation Actions

1. Protect roost sites from mining, human disturbance and collapse.
2. Replace the top strands of barbed wire in fences near roost sites with single-strand wire.

Conservation and Management Actions

The majority of known colonies occur in protected areas (e.g. national parks or heritage listed mine sites) (McKenzie & Hall 2008). However, some breeding sites, for example in the Pilbara, are not protected and no formal monitoring plan has been implemented (Armstrong & Anstee 2000; K. Armstrong pers. comm., cited in McKenzie & Hall 2008). Current management activities include protection of some breeding sites, a captive breeding programme, long-term population studies and monitoring in Queensland, and population studies in Western Australia (McKenzie & Hall 2008; WA DPaW 2015).

Bullen (pers. comm., 2015) notes that while some roosting sites are protected, extended habitat retention at ridge and creek line scales surrounding roosting sites is needed, as well as protection of these areas from disturbance (including from airborne dust clouds which affect the bats' eyesight and hunting success, and burying of preferred foraging habitat under stored overburden).

Recommended management actions are outlined in the table below (Woinarski et al., 2014).

Theme	Specific actions	Priority
Active mitigation of threats	Protect land with significant colonies.	High
	In barbed wire fences close to roost sites, replace the top strand with single-strand wire, and put a metal disc (around 10x10cm) between the top and second strands.	High
	Protect roost sites and surrounding foraging areas from disturbance, including the loss of habitat quality due to changes to fire and grazing regimes.	Medium
	Where appropriate, modify roost site areas to reduce risks of collapse, and ensure mine-adits that are known roost sites for ghost bats are maintained following the cessation of mining activities.	Medium
Captive breeding	N/a	
Quarantining isolated populations	N/a	
Translocation	N/a	
Community engagement	Educate people not to disturb roost sites.	Medium
Reduce disturbance of roost sites	Where there are known roosts in proximity to mining or other activities, ensure disturbance is minimised by undertaking environmental assessment, considering alternative locations for works and impact mitigation measures.	High

Survey and monitoring priorities

Theme	Specific actions	Priority
Survey to better define distribution	Collate and review all information on Pilbara roost sites, and identify banded-ironstone areas in all parts of the region that are planned for future mining or may be quarantined from mining.	High
	Additional surveys, especially to locate breeding sites, are required in remote parts of the Pilbara, Kimberley and Northern Territory.	High
	Assess population size (and significance) of all known subpopulations.	Medium-high
Establish or enhance monitoring program	Monitor populations at key sites and where impacts from mining are occurring or likely.	High

	Develop cost-effective monitoring protocols (e.g. thermal tracking software) at a set of standardised sites that contain most of the known population.	Medium

Information and research priorities

Theme	Specific actions	Priority
Assess impacts of threats on species	Assess impacts of disturbance of breeding sites, and identify appropriate buffer zones for specific activities around roost sites so mining and other activities do not lead to abandonment.	High
Assess effectiveness of threat mitigation options	Assess options for establishment of new/artificial roost sites (as a last resort only), and mitigation options to reduce impacts of mining. Evaluate the success of such actions.	Medium
Resolve taxonomic uncertainties	N/a	
Assess habitat requirements	Assess seasonal access to foraging areas in the Pilbara remote from major roosts.	Medium
Assess diet, life history	Assess proximity to roosts of foraging habitats used by lactating females compared to other adults.	Medium

Recommendations

(i) The Committee recommends that the list referred to in section 178 of the EPBC Act be amended by **including** in the list in the Vulnerable category:

Macroderma gigas

(ii) The Committee recommends that there should be a recovery plan for this species.

Threatened Species Scientific Committee

2/3/2016

References cited in the advice

Armstrong, K. N. (2001). The roost habitat and distribution of the orange leaf-nosed bat, *Rhinonicteris aurantius*, in the Pilbara region of Western Australia. *Wildlife Research 28*, 95-104.

Armstrong, K. N. (2010). Assessing the short-term effect of minerals exploration drilling on colonies of bats of conservation significance: a case study near Marble Bar, Western Australia*. Journal of the Royal Society of Western Australia 93*, 165-174.

Armstrong, K. N. (2011). The current status of bats in Western Australia. In *The Biology and Conservation of Australasian Bats* (eds B. Law, P. Eby, D. Lunney & L. Lumsden), pp. 257-269. Royal Zoological Society of New South Wales, Mosman.

Armstrong, K. N., & Anstee, S. D. (2000). The ghost bat in the Pilbara: 100 years on. *Australian Mammalogy 22*, 93-101.

Armstrong, K. N., Chavand, O. & Worthington-Wilmer, J. (In prep). Population genetic assessment of the Pilbara Ghost Bat *Macroderma gigas* (Chiroptera: Megadermatidae).

Augusteyn, J.D. Hughes, J. Armstrong, G. & Real, K. (In prep). Tracking and tracing Central Queensland's Macroderma - determining the size of the Mt Etna ghost bat population and its key foraging habitat. *Wildlife Research x*, xx-xx.

Boles, W. E. (1999). Avian prey of the Australian Ghost Bat *Macroderma gigas* (Microchiroptera: Megadermatidae): prey characteristics and damage from predation. *Australian Zoologist 31*, 82-91.

Burbidge, A. A., Johnson, K. A., Fuller, P. J., & Southgate, R. I. (1988). Aboriginal knowledge of the mammals of the central deserts of Australia. *Australian Wildlife Research 15*, 9-39.

Burbidge, A. A., McKenzie, N. L., Brennan, K. E. C., Woinarski, J. C. Z., Dickman, C. R., Baynes, A., Gordon, G., Menkhorst, P. W., & Robinson, A. C. (2009). Conservation status and biogeography of Australia's terrestrial mammals. *Australian Journal of Zoology 56*, 411-422.

Butler, W. H. (1962). Occurrence of the Ghost Bat, *Macroderma gigas*, in the Great Victoria Desert, W.A. *The Western Australian Naturalist 8*, 42-43.

Churchill, S. K. (1991). Distribution, abundance and roost selection of the Orange Horseshoe-bat, *Rhinonycteris aurantius*, a tropical cave dweller. *Wildlife Research 18*, 343-353.

Churchill, S. K., & Helman P. M. (1990). Distribution of the ghost bat, *Macroderma gigas*, (Chiroptera: Megadermatidae) in central and south Australia. *Australian Mammalogy 13*, 149-156.

Department of the Environment (DotE) (2016). Area of Occupancy and Extent of Occurrence for *Macroderma gigas*. Unpublished report, Australian Government Department of the Environment, Canberra.

Douglas, A. M. (1962). *Macroderma gigas saturata* (Chiroptera, Megadermatidae) a new subspecies from the Kimberley Division of Western Australia. *The Western Australian Naturalist 8*, 59-61.

Duncan, A., Baker, G.B. & Montgomery, N. (1999). *The Action Plan for Australian Bats*. Biodiversity Group, Environment Australia, Canberra.

Filewood, L. W. (1983). The possible occurrence in New Guinea of the Ghost Bat (*Macroderma gigas*; Chiroptera, Megadermatidae). *Australian Mammalogy 6*, 35-36.

Grant, C., Reardon, T., & Milne, D. (2010). Ghost Bat count at Kohinoor Adit. *Australasian Bat Society Newsletter no. 35*, 36-38.

Hall, L., Richards, G., McKenzie, N., & Dunlop, N. (1997). The importance of abandoned mines as habitat for bats. In *Conservation Outside Nature Reserves* (eds P. Hales & D. Lamb), pp. 326-333. The University of Queensland, Brisbane.

Hand, S., & York, A. (1990). Morphological variation in the dentition and skull of the Australian Ghost Bat, *Macroderma gigas* (Microchiroptera, Megadermatidae). *Australian Journal of Zoology 38*, 263-286.

Hoyle, S. D., Pople, A. R., & Toop G. J. (2001). Mark–recapture may reveal more about ecology than about population trends: Demography of a threatened ghost bat (*Macroderma gigas*) population. *Austral Ecology 26*, 80-92.

Hutson, A. M., Mickleburgh, S. P. & Racey, P. A. (2001). *Microchiropteran Bats - Global Status Survey and Conservation Action Plan*. IUCN/SSC Chiroptera Specialist Group, Gland, Switzerland and Cambridge, U.K.Koopman, K. (1984). Taxonomic and distributional notes on tropical Australian bats. *American Museum Novitates 2778*, 1-48.

Leitner P., & Nelson, J. E. (1967). Body temperature, oxygen consumption and heart rate in the Australian False vampire Bat *Macroderma gigas*. *Comparative Biochemistry and Physiology 21*, 65-74.

McKenzie, N. L., & Bullen, R. D. (2009). The echolocation calls, habitat relationships, foraging niches and communities of Pilbara microbats. *Records of the Western Australian Museum* Supplement *78*, 123-155.

McKenzie, N. L., & Bullen, R. D. (2012). An acoustic survey of zoophagic bats on islands in the Kimberley, Western Australia, including data on the echolocation ecology, organisation and habitat relationships of regional communities. *Records of the Western Australian Museum Supplement 81*, 67-107.

Milne, D.J. & Pavey, C. R. (2011). The status and conservation of bats in the Northern Territory. In *The biology and Conservation of Australasian Bats* (eds B. Law, P. Eby, D. Lunney & L. Lumsden), pp. 208-225. Royal Zoological Society of New South Wales, Mosman.

Molnar R. E., Hall, L. S., & Mahoney J. H. (1984). New fossil localities for *Macroderma* Miller, 1906 (Chiroptera: Megadermatidae) in New South Wales and its past and present distribution in Australia. *Australian Mammalogy 7*, 63-73.

Pettigrew, J., Baker, G. B., Baker-Gabb, D., Baverstock, G., Coles, R., Conoloe, L., Churchill, S., Fitzherbert, K., Guppy, A., Hall, L., Helman, P., Nelson, J., Priddel, D., Pulsford, I., Richards, G., Schulz, M., & Tidemann, C. R. (1986). The Australian Ghost Bat at Pine Creek, Northern Territory. *Macroderma 2*, 8-19.

Reardon, T. B., Robson, S. K. A., Parsons, J. G., & Inkster, T. (2010). Review of the threatened status of microchiropteran bat species on Cape York Peninsula. South Australian Museum, Adelaide.

Richards, G. C., Hand, S., Armstrong, K. A., & Hall, L. S. (2008). Ghost Bat *Macroderma gigas*. In *The Mammals of Australia*. Third Edition. (Eds S. Van Dyck & R. Strahan), pp. 449-450. Reed New Holland, Chatswood.

Schulz, M. (1986). Vertebrate prey of the ghost bat, *Macroderma gigas*, at Pine Creek, Northern Territory. *Macroderma 2*, 59-62.

Simmons N. B. (2005). Order Chiroptera. *In Mammal Species of the World: a taxonomic and geographic reference*. Third edition. (Eds D. E. Wilson & D. M. Reeder), pp. 312-529. Johns Hopkins University Press, Baltimore.

Thomson, V., Armstrong, K. N, Medlin, G., & Cooper, A. (2012). Ghost of a chance? Evolutionary history of the Ghost Bat (*Macroderma gigas*) and its chances of surviving future climate change. *Australian Mammal Society 58th Scientific Meeting Abstracts*.

Tidemann, C. R., Priddel, D. M., Nelson, J. E., & Pettigrew, J. D. (1985). Foraging behaviour of the Australian Ghost Bat, *Macroderma gigas* (Microchiroptera: Megadermatidae). *Australian Journal of Zoology 33*, 705-713.

Toop, G. J. (1985). Habitat requirements, survival strategies and ecology of the ghost bat, *Macroderma gigas* Dobson (Microchiroptera, Megadermatidae) in central coastal Queensland. *Macroderma 1*, 37-41.

White, A.W., Morris, I., & G. Madani. (in prep). Ghost Bat Declines in Northern Australia. Are Cane Toads *Rhinella marina* implicated?

Woinarski, J. C. Z., Burbidge, A. A., & Harrison, P. L. (2014). *The Action Plan for Australian Mammals 2012*. CSIRO Publishing, Collingwood.

Worthington Wilmer, J. (1996). Genetic variation and population structure in the threatened Ghost Bat (*Macroderma gigas*). Ph.D. Thesis, University of Queensland, Brisbane.

Worthington Wilmer, J. (2012). Ghost Bat *Macroderma gigas*. In *Queensland's Threatened Animals* (eds L. K. Curtis, A. J. Dennis, K. R. McDonald, P. M. Kyne & S. J. S. Debus), pp. 382-383. CSIRO, Canberra.

Worthington Wilmer, J., Moritz, C., Hall, L., & Toop, J. (1994). Extreme population structuring in the threatened ghost bat, *Macroderma gigas*: evidence from mitochondrial DNA. *Proceedings of the Royal Society of London B Biological Sciences 257*, 193-198.

Worthington Wilmer, J., Hall, L., Barratt, E., & Moritz, C. (1999). Genetic structure and male-mediated gene flow in the Ghost Bat (*Macroderma gigas*). *Evolution 53*, 1582-1591.

Other sources cited in the advice

Australian Wildlife Conservancy (2010). *Wildlife Matters.* Winter 2010. Available on the internet at: https://www.australianwildlife.org/media/104804/AWC-Wildlife-Matters-Winter-2010.pdf.

AnAge (2012). The animal aging and longevity database. Available on the internet at: http://genomics.senescence.info/species/

Armstrong, K. N. (2015). Personal communication by email, 25 November 2015. Zoologist and President of the Australasian Bat Society.

Bullen, B. (2015). Personal communication by email, 5 November 2015. Ecologist and Principal of Bat Call WA Pty Ltd.

Hughes, T. (2015). Personal communication by email, 2 October 2015. Director, South Endeavour Trust, owners of Kings Plains Station.

McKenzie, N., & Hall, L. (2008). *Macroderma gigas*. In 'IUCN red list of threatened species.' Version 2012.1. Viewed 1 July 2012. Available on the internet at: http://www.iucnredlist.org.

Queensland Department of Environment and Heritage Protection (Qld DEHP) (2015). Submission on the listing assessments to assist the Threatened Species Scientific Committee. Received 25 November 2015.

Toop, J., & Davies, J. (unpublished). Conservation of the ghost bat *Macroderma gigas* in central Queensland. QPWS Rockhampton.

Western Australia Department of Parks and Wildlife (WA DPaW) (2015). Submission on the nomination for listing *Macroderma gigas* (ghost bat). Received 25 November 2015.

Whilte, A. (2016). Personal communication by email, 2 March 2016. Biosphere Environmental Consultants Pty Ltd.

White, N.J. (2015a). Personal communication by email, 25 November 2015. Chair, Conservation Commission, Australian Speleological Federation Inc.

White, N.J. (2015b). Personal communication by email, 2 March 2016. Chair, Conservation Commission, Australian Speleological Federation Inc.

Australian Government
Department of Climate Change, Energy, the Environment and Water

Conservation Advice for *Petauroides minor* (greater glider (northern))

In effect under the *Environment Protection and Biodiversity Conservation Act 1999* from 5 July 2022.

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

Greater glider (northern) © Copyright, Carly Starr (from The Bluff State Forest, Tumoulin N Qld)

Conservation status

Petauroides volans (greater glider) 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 5 May 2016.

This assessment recognises that *P. volans*, as understood in 2016 is now considered to be at least two separate species: *P. volans* (greater glider (southern and central)) and *P. minor* (greater glider (northern)) (McGregor et al. 2020).

Petauroides minor (northern) 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 assessment of the species' eligibility against each of the listing criteria is:

- Criterion 1: A2c+3c+4c: Vulnerable
- Criterion 2: B2ab(i,ii,iii,iv,v): Vulnerable
- Criterion 3: Not eligible
- Criterion 4: Not eligible
- Criterion 5: Insufficient data

The main factors that make the species eligible for listing in the Vulnerable category are population reduction, limited area of occupancy, and continuing decline in the area and quality of habitat primarily due to 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.

The current listing status of this species under the Queensland (Qld) *Nature Conservation Act 1992* is Vulnerable since October 2014. *Petauroides minor* (greater glider (northern)) and *Petauroides volans* (greater glider (southern)) are both included in the listing.

Species information

Taxonomy

Conventionally accepted as *Petauroides minor* Collett (1887).

Formerly, *Petauroides volans* was the only species in the genus. Two subspecies were recognised: *P. v. minor* (in north-eastern Qld) and *P. v. volans* (in south-eastern Australia) (van Dyck & Strahan 2008).

Jackson & Groves (2015) split the species into three separate species: *P. minor* (Atherton Tablelands and coastal central and northern Qld), *P. armillatus* (inland central Qld), and *P. volans* (from south-east Qld to Victoria (Vic)). McGregor et al. (2020) agreed with this taxonomic arrangement within *Petauroides* on the basis of genomic-scale nuclear markers and external morphological data.

A new dataset that combined the genetic resources of McGregor et al. (2020) and that of B. Arbogast & K. Armstrong et al. (manuscript in prep.), which included more extensive sampling throughout the range of *Petauroides* for genomic-scale markers, a mitochondrial marker dataset and cranial measurements, has supported the separate recognition of *P. minor* (K.N. Armstrong 2021. pers comm 24 June).

Therefore, the listed entity in this Conservation Advice is referred to as *Petauroides minor* (greater glider (northern)), while the common name greater glider refers to the genus *Petauroides.*

Description

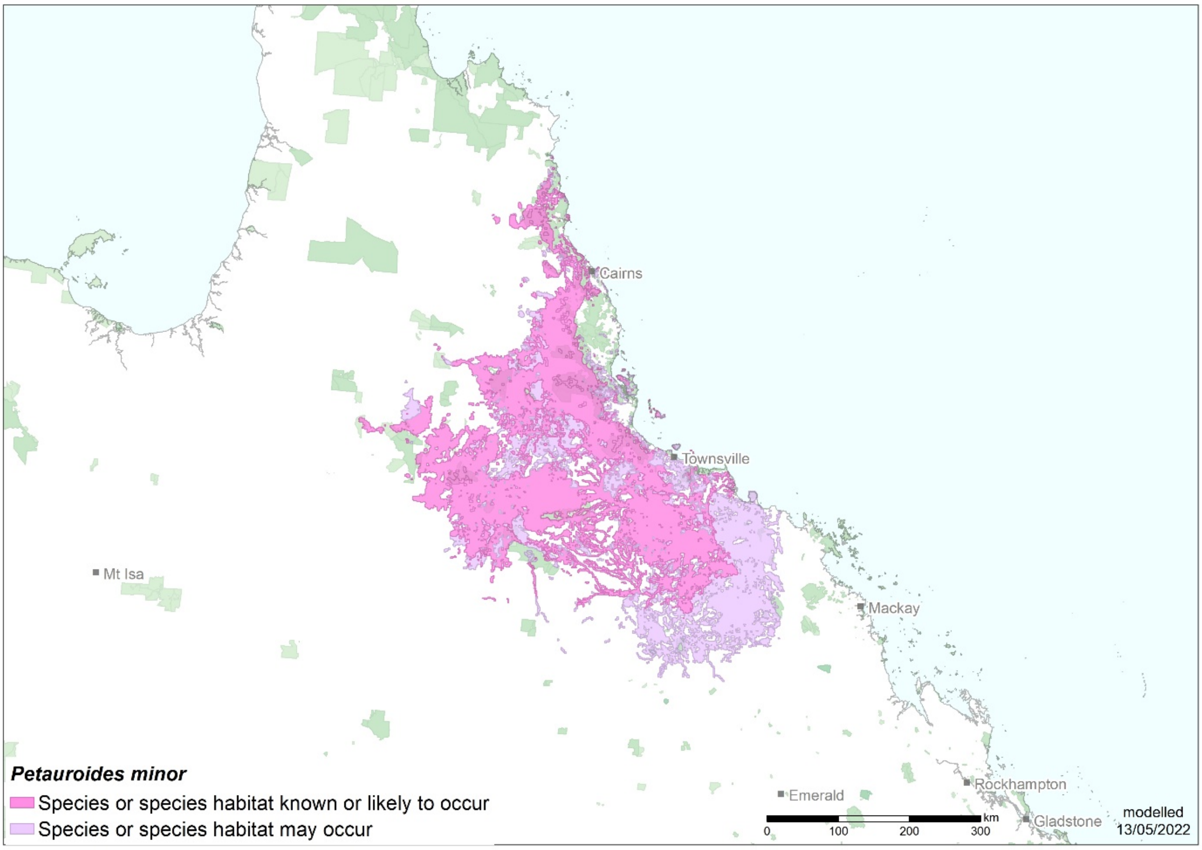
The greater glider (northern) is the largest gliding possum in north-eastern Australia. It has a head and body length of 32−40 cm, tail length of 40−48 cm, and a weight range of 650−1100 g, with females being larger than males (McKay 1989, 2008; McGregor et al. 2020). The greater glider (northern) has thick fur that increases its apparent size. Its fur colour is dusky brown above, often with a darker mid-dorsal stripe, and whitish below. It has a more slender body, with shorter ears and tail, than the greater glider (southern and central) (Comport et al. 1996). Its tail is not prehensile, and the gliding membrane extends from the forearm to the tibia (McKay 1989, 2008).

Distribution

The greater glider (northern) occurs in the wet-dry tropical region of north-eastern Australia, including the Wet Tropics World Heritage Area. It is distributed from around Townsville northwards to the Windsor Tablelands (McGregor et al. 2020; B Arbogast & KN Armstrong et al. unpublished data; OZCAM records: Atlas of Living Australia 2021). This distribution is very patchy with some isolated subpopulations, for example in the Gregory Range/Gilbert Plateau west of Townsville (Winter et al. 2004) and Blackbraes National Park (Vanderduys et al. 2012).

The broad extent of occurrence (EOO) is unlikely to have changed appreciably since European settlement (Woinarski et al. 2014). However, the area of occupancy (AOO) has decreased substantially, mostly due to land clearing. This area is probably continuing to decline due to further clearing, fragmentation impacts, edge effects, bushfire, climate change and some forestry activities (Woinarski et al. 2014). Kearney et al. (2010) predicted a 'stark' and 'dire' decline of suitable habitat ('almost complete loss' ~ 90 percent) for the greater glider (northern) if there is a 3 °C temperature increase.

Map 1 Modelled distribution of greater glider (northern)



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 greater glider (northern) is poorly known. However, the habitats and area in which the greater glider (northern) are found have a long and profound history of management by Indigenous Australians.

Relevant biology and ecology

General habitat
The greater glider (northern) is an arboreal nocturnal marsupial, predominantly solitary and largely restricted to eucalypt forests and woodlands of north-eastern Australia. It is typically found in highest abundance on high elevation, wetter sites in open woodland to open forests, containing relatively old trees and abundant hollows (Eyre 2004; Vanderduys et al. 2012). It is likely that only a proportion of forest in potential habitat areas is suitable for the species, as the structural attributes of the forest overstorey and forage quality it relies on vary considerably across the landscape (Eyre 2002; Youngentob et al. 2011).

Den trees
During the day the greater glider (northern) shelters in tree hollows, with a particular preference for large hollows (diameter >10 cm) in large, old trees (Kehl & Borsboom 1984; Smith et al. 2007; Goldingay 2012). Comport et al. (1996) reported that *Eucalyptus acmenoides* (white mahogany) and *Corymbia citriodora* (lemon-scented gum) were the favoured denning trees for the greater glider (northern), and the species utilised 4–6 dens per month. In the north of its range *E. tereticornis* (forest red gum) is favoured for denning, and two dens per hectare are utilised (Starr et al. 2021).

Diet
It is primarily folivorous, with a distinct preference for young foliage (Comport et al. 1996), supplemented by buds and flowers. It feeds from a restricted range of eucalypt species, and favours forests with a diversity of eucalypt species due to seasonal variation in its preferred tree species (Comport et al. 1996). The tree species favoured by greater gliders varies regionally. Approximately 85% of the greater glider's water requirements are provided by consumed leaves (Foley et al. 1990). Free water is presumably obtained from dew condensation on leaf surfaces (Rübsamen et al. 1984).

Life history
The greater glider's (northern) life history is assumed to be similar to the greater glider's (southern and central), where females give birth to a single young from March to June (Tyndale-Biscoe & Smith 1969b; McKay 2008). Sexual maturity is reached in the second year (Tyndale-Biscoe & Smith 1969b). Longevity has been estimated at 15 years (Jones et al. 2009), and generation length is estimated to be six to eight years (Pacifi et al. 2013; Woinarski et al. 2014). The relatively low reproductive rate (Henry 1984) may render small subpopulations in isolated remnants prone to extinction (van der Ree 2004; Pope et al. 2004).

Home ranges and densities
Home ranges are typically relatively small and are larger for males (2.5 ha) than for females (1.3 ha), with male home ranges being overlapping with other males and females (Comport et al. 1996), indicating a polygamous mating system. Starr et al. (2021) reported that the greater glider (northern) has a home range of about 1–12 ha, with home ranges also overlapping.

The density of the greater glider (northern) has been reported as 3.3–3.8 ha^{-1} for Taravale Station north-west of Townsville (Comport et al. 1996), 2.6–5.8 ha^{-1} for Blackbraes National Park (Vanderduys et al. 2012), and recently in the north of its range (the Bluff State Forest) as 0.24 to 0.38 individuals per hectare in wet and dry sclerophyll forest respectively (Starr et al. 2021). Vanderduys et al. (2012) reported the greater glider (northern) at high densities from two land zones in the Einasleigh Uplands, which were described as 1) *E. crebra* (narrow-leaved ironbark) and/or *Eucalyptus* spp. and/or *Corymbia* spp. open woodland to open forest on gently undulating sandplain plateaus; and 2) *Eucalyptus* spp., lemon-scented gum and white mahogany open forest on high plateaus on earths and sands.

Disturbance ecology
While there is very little available information regarding the effect of disturbance on the greater glider (northern), its similar ecology and biology means its responses to disturbance can be expected to be very similar to the greater glider (southern and central). The greater glider (southern and central) is particularly sensitive to forest clearance (Tyndale-Biscoe & Smith 1969a) and to intensive logging (Kavanagh & Bamkin 1995; Kavanagh & Webb 1998; Kavanagh & Wheeler 2004; Mclean et al. 2018; Starr et al. 2021). Responses vary according to landscape context and the intensity of disturbance (Kavanagh 2000; Taylor et al. 2007). Greater glider populations are slow to recover following major fires (Kavanagh 2004) due to the low reproductive rate of the species and its limited dispersal capabilities. Substantial losses or declines of greater glider populations have been documented after fires, through direct mortality and indirect impacts on habitat (McLean et al. 2018).

The greater glider (northern) is likely sensitive to fragmentation, similarly to the greater glider (southern and central) (McCarthy & Lindenmayer 1999a,b; Lindenmayer et al. 2000; Eyre 2006; Taylor & Goldingay 2009) which has relatively low persistence in small forest fragments, and disperses poorly across vegetation that is not native forest (Pope et al. 2004).

Habitat critical to the survival

Habitat critical to survival for the greater glider (northern) may be broadly defined as (noting that geographic areas containing habitat critical to survival needs to be defined by forest type on a regional basis):

- large contiguous areas of eucalypt forest, which contain mature hollow-bearing trees[1] and a diverse range of the species' preferred food species in a particular region; and
- smaller or fragmented habitat patches connected to larger patches of habitat, that can facilitate dispersal of the species and/or that enable recolonization; and
- cool microclimate forest/woodland areas (e.g. protected gullies, sheltered high elevation areas, coastal lowland areas, southern slopes); and
- areas identified as refuges under future climate changes scenarios; and
- short-term 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.

[1] Tree hollows can be difficult to detect in ground-based surveys. The presence of trees with basal diameter > 30 cm can be used as a proxy measure for tree hollows used by greater gliders in Queensland (Eyre et al. 2021).

Habitat meeting any one of the criteria above is considered habitat critical to the survival of greater glider (northern), irrespective of the current abundance or density of greater gliders or the perceived quality of the site. Forest areas currently unoccupied by the greater glider (northern) 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 subpopulation.

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.

The number and locations of populations and metapopulations of the greater glider (northern) across its distribution have not been determined. However, an important population may be defined as a population that occurs:

- in a defined geographical area containing habitat critical to survival; or
- in areas where the species persists in relatively higher density or abundance at a regional level; or
- where its habitat provides refugia in times of stress or from threatening processes (particularly where other nearby populations have substantially declined or may be expected to do so in the future); or
- populations that are isolated or occur at the margins of the species' range, that may be important for maintaining genetic diversity and evolutionary adaptation.

There are two known isolated populations, one in the Gregory Range/Gilbert Plateau west of Townsville, and one in the Einasleigh uplands (Winter et al. 2004; Vanderduys et al. 2012). These isolated populations should be considered to be important populations.

Threats

Key threats to the greater glider (northern) are climate change, land clearing and timber harvesting (Table 1). There are synergies between these threats, and their combined impact needs to be considered in the recovery of the species. Loss and fragmentation of habitat has already occurred in many areas of the species' range (Woinarski et al. 2014), and the impacts of climate change will place increased pressure on its remaining habitat.

Table 1 Threats impacting the greater glider (northern)

Threat	Status and severity [a]	Evidence
Climate Change		
Increased temperatures and changes to rainfall patterns	• Timing: current and future • Confidence: inferred • Consequence: catastrophic • Trend: increasing • Extent: across the entire range	The greater glider's unique physiology and a strict eucalypt diet make it vulnerable to high temperatures and low water availability (Rübsamen et al. 1984). Prolonged exposure to temperatures over 40°C is likely to lead to high mortality (Rübsamen et al. 1984). Moore et al. (2004) suggested that the preference of greater gliders for higher elevations is because they are sensitive to heat and must expend energy and considerable water to cool themselves when the ambient temperature is over 20°C. Climate change projections show that the Wet Tropics and monsoonal north-east of Australia will experience increases in average and maximum temperatures, frequency of hot days, the duration of warm spells, and intensity of extreme rainfall events (McInnes et al. 2015; Moise et al. 2015). Biophysical modelling predicts a severe and dire range contraction (~90%) for the greater glider (northern), under a 3 °C temperature increase (Kearney et al. 2010). A warmer climate also reduces the nutritional and water content of eucalypt leaves (Foley et al. 1990; Lawler et al. 1997; Gleadow et al. 1998; McKiernan et al. 2014). While changes in the amount of rainfall in the distribution of the greater glider (northern) are uncertain under future climate change projections (McInnes et al. 2015; Moise et al. 2015), a warmer climate is likely to impact food availability for greater gliders and could be expected to reduce reproduction rate and population size (DeGabriel et al. 2009; Kearney et al. 2010). At high temperatures greater gliders reduce their food intake due to thermogenesis, leading to their energy and water stores being rapidly expended (Beale et al. 2018; Youngentob et al. 2021). Above temperatures of 35°C, greater gliders need to dissipate >100% of metabolic heat production by evaporative means (Rübasamen 1984). This can lead to death of both young and adult gliders, or if less severe, can reduce growth in milk-fed young and reduce the health and fitness of adult gliders (Youngentob et al. 2021). There are limited documented responses of the greater glider (northern) to increased temperatures and changes to rainfall patterns. However, responses are likely to be similar to that of the greater glider (southern and central), for which the following has been documented: higher night-time temperatures were attributed to be the cause of population declines at lower altitude (<500 m) surveyed sites in the Blue Mountains (Smith & Smith 2018, 2020); increased night-time temperatures have been implicated in the declines of some Victorian subpopulations (Wagner et al. 2020).

Threat	Status and severity [a]	Evidence
Habitat disturbance and modification		
Habitat clearing and fragmentation	• Timing: current and future • Confidence: observed • Consequence: catastrophic • Trend: unknown • Extent: across parts of the range	Much of the woodland and open forest habitat in the greater glider's (northern) range has been cleared, primarily due to development, agriculture and timber production (Woinarski et al. 2014). Extensive clearing and habitat degradation is continuing (Woinarski et al. 2014). The greater glider (northern) is absent from cleared areas and has little dispersal ability to move through cleared areas between fragments (Comport et al. 1996). Fragmentation effects are likely exacerbated by inappropriate fire regimes.
Timber harvesting	• Timing: current and future • Confidence: observed • Consequence: major • Trend: unknown • Extent: across parts of the range	Timber harvesting occurs in some habitat in the species' range (Woinarski et al. 2014). The greater glider (northern) is highly dependent on forest connectivity and large mature trees, and impacts are similar to the greater glider (southern and central). Fire-logging interactions likely increase risks to greater glider populations.
Inappropriate fire regimes	• Timing: current and future • Confidence: inferred • Consequence: moderate • Trend: unknown • Extent: across parts of the range	Responses to fire have not been documented for this species. However, its responses to fire are likely to be similar to the greater glider (southern and central). No known bushfire events have substantially impacted the greater glider (northern). The species was only minimally impacted by the unprecedented 2019-20 bushfires in south-eastern Australia, which overlapped an estimated 0.1% of the species' distribution (Legge et al. 2021). However, it is possible that bushfires could become a greater threat to the species in the future due to climate change. Altered weather conditions are leading to higher frequency and intensity of bushfires (CSIRO 2020). Although there are limited documented responses of the greater glider (northern) to increased frequency and intensity of bushfires, its response is likely to be similar to that of the greater glider (southern), for which substantial population losses or declines have been documented in and after high intensity fires (Lindenmayer et al. 2013; Berry et al. 2015; McLean et al. 2018). Conversely, vegetation change is occurring in some parts of its range due to reduced fire frequency and intensity, which has resulted in rainforest encroachment on wet sclerophyll forest (Harrington & Sanderson 1994; Winter et al. 2004). Sclerophyll trees are unable to regenerate in shade and usually require fire to provide the appropriate conditions (Harrington & Sanderson 1994). This altered vegetation structure and floristics may reduce habitat suitability for the greater glider (northern), including the availability of hollows and food trees (Winter et al. 2004).
Barbed wire fencing (entanglement)	• Timing: current and future • Confidence: observed • Consequence: minor • Trend: unknown • Extent: across the entire range	There are occasional losses of individuals due to entanglement in barbed wire fences across the greater glider's range (van der Ree 1999).

Threat	Status and severity [a]	Evidence
Introduced species		
Predation by feral cats (*Felis catus*)	• Status: current and future • Confidence: observed • Consequence: minor • Trend: unknown • Extent: across parts of the range	Remains of greater gliders have been found in the stomachs of feral cats, however they formed a tiny proportion of the overall animals consumed (Jones & Coman 1981). It is unclear whether they were killed by cats (if so, most likely when gliders come to the ground) or consumed as carrion. After wildfires, greater gliders are displaced and have been observed on the ground where they are more susceptible to predation (Fleay 1947), suggesting that fire-predator interactions amplify threats to the species.

Timing—identify the temporal nature of the threat;
Confidence—identify the extent to which we have confidence about the impact of the threat on the species;
Consequence—identify the severity of the threat;
Trend—identify the extent to which it will continue to operate on the species;
Extent—identify its spatial content in terms of the range of the species.

Each threat has been described in Table 1 in terms of the extent that it is operating on the species. 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 Greater glider (northern) risk matrix

Likelihood	Consequences				
	Not significant	**Minor**	**Moderate**	**Major**	**Catastrophic**
Almost certain	Low risk	Moderate risk	Very high risk	Very high risk **Timber harvesting**	Very high risk **Increased temperatures and changes to rainfall patterns** **Habitat clearing and fragmentation**
Likely	Low risk	Moderate risk	High risk	Very high risk	Very high risk
Possible	Low risk	Moderate risk	High risk **Inappropriate fire regimes**	Very high risk	Very high risk
Unlikely	Low risk	Low risk **Barbed wire fencing (entanglement)** **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

Within the next three generations, the population size as well as the extent, quality and connectivity of habitat required to maintain the population will have increased.

Conservation and management priorities

Climate change

- Protect all habitat likely to be climate change refuges, including sites buffered against desiccating conditions (e.g. sheltered and/or on south-facing aspects), under future climate change scenarios. Where possible, maintain or establish connectivity with existing habitat in order to facilitate movement.
- Where feasible, undertake habitat restoration/enhancement to improve micro-climate conditions in areas at high risk of extreme temperatures and drought.
- Ensure that eucalypt forests and the impacts of disturbance are managed to prevent them transitioning to less nutritious, hotter, and/or more fire-prone plant communities, and to ensure that food tree species preferred by the greater glider (northern) continue to be the dominant canopy trees.

Habitat loss, disturbance and modification (including fire)

- Protect and maintain sufficient areas of suitable habitat, including denning and foraging resources and habitat connectivity, to sustain viable populations throughout the species' range.
- In the aftermath of bushfires, protect any unburnt habitat (within or adjacent to recently burnt landscapes) in order to support population recovery. This includes, but is not limited to:
    - Areas identified to be important post-fire refuges.
    - Protecting hollow-bearing trees from post-fire salvage timber harvesting and clean-up operations.
    - Avoiding hazard reduction burns in these areas.

- Re-assess and revise current prescriptions used for prescribed burning to ensure that the frequency and severity of fires in greater glider habitat are minimised, in order to mitigate the risk of further population declines and loss of hollow-bearing trees. Measures to reduce risk from future bushfires should be strategic, incorporate adaptive management, and include a risk assessment that considers trade-offs between fire control efficiency and environmental damage.
- Implement and enforce measures to reduce direct mortality and loss of hollow-bearing trees during site preparation and execution of prescribed burns, including rake hoeing around the base of trees.
- Protect hollow-bearing trees on private property, roadside reserves, and along the edges of roads/tracks. Prior to removing trees identified to be a 'hazard', undertake a risk assessment by a suitably qualified person to determine whether their removal is necessary, including a consideration of the potential impacts of tree removal on the greater glider. Incorporate measures to ensure ongoing recruitment of hollow-bearing trees into planning processes.
- Avoid fragmentation and loss of habitat due to development of new transport corridors. Incorporate avoidance and protection measures for the species into planning processes, and where possible re-locate recreational activities and roads away from habitat.
- Establish, maintain and enforce effective prescriptions in production forests to support subpopulations of the greater glider (northern). This includes, but is not limited to: appropriate levels of habitat retention, timber harvesting exclusion and timber harvesting rotation cycles; maintenance of wildlife corridors between harvested patches; maintenance of vegetation buffers around habitat patches excluded from harvesting; protection of existing hollow-bearing trees with appropriate buffers; adequate recruitment of hollow-bearing trees; maintaining preferred food tree species as dominant canopy trees; and minimal use and adequate containment of regeneration burns. Timber harvesting in climate or post-fire refuges should be avoided.
- As a last resort, where hollows are limiting, consider the use of nest boxes and artificial hollows that are suitable for the species. Monitor use of these structures to ensure they are being utilised, and revise designs or placement as required.
- Restore habitat and connectivity:
    - where habitat has been substantially fragmented, disturbed or modified,
    - between small habitat patches and larger areas of contiguous forest,
    - at a landscape scale, to facilitate movement and recolonisation of areas impacted by fires, droughts or other factors, and to provide opportunities for the species to adapt to the changing climate,
    - following climate-ready restoration guidelines (e.g. Hancock et al. 2018), and
    - following the National Restoration Standards (Standards Reference Group SERA 2021).
- Revise mitigation and offset guidelines for development and linear infrastructure (e.g. pipelines, transport corridors) to reflect the limited effectiveness of artificial structures (nest boxes, glide poles) as mitigation actions for loss, degradation or fragmentation of greater glider habitat.

- Avoid the use of barbed wire, and replace the top strands of existing barbed wire with single-strand wire in habitat known to be occupied by greater gliders.

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

- Where threats from introduced predators (including the feral cat) are locally significant:
    - Implement appropriate control measures, particularly in areas burnt by bushfires.
    - Develop and implement longer-term strategies to control predation by the feral cat, as detailed in the relevant Threat Abatement Plan.

Stakeholder engagement/community engagement

- Seek stakeholder input into assessment and planning processes that include protections for the greater glider (northern) and its habitat. This may include environmental impact assessments, park management plans, water resource plans, fire management plans and transport development plans.
- Develop and implement a communication strategy around the need to balance hazard reduction burning with the need to conserve and protect species and habitats.
- Liaise with private landholders, Traditional Owners, and conservation and land management groups to create guidelines for on-ground management of the greater glider (northern).
- Support volunteer involvement in surveying and monitoring, in particular gathering data on the species' occurrence and foraging habitat, and in the implementation of conservation actions.
- 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 species.
- Engage and involve Traditional Owners in conservation actions, including survey, monitoring and management actions.
- Foster public interest in the species and its ongoing conservation, to increase support for the implementation of conservation actions.

Survey and monitoring priorities

- Implement an integrated long-term monitoring program across the species' range to:
    - determine trends in abundance and distribution,
    - ascertain the status and viability of subpopulations,
    - assess the impacts of (individual and compounding) threats, and
    - evaluate the relative benefits and effectiveness of management actions.
- Following disturbance events such as bushfires, heatwaves or drought, conduct on-ground surveys to establish habitat and population impacts as a result of the event and to provide a baseline for future population monitoring. Leverage post-disturbance monitoring at sites where surveys were undertaken prior to the event, to assess population trends.
- Monitor the incidence and impacts of fire and timber harvesting in the species' range.

- Monitor the abundance, age and size structure of hollow-bearing trees and their responses to management measures. This includes before and after prescribed burns, and before and after timber harvesting.
- Undertake surveys on high priority timber harvesting coupes, and other pre-harvest surveys, to inform adaptive management in timber harvesting areas.

Information and research priorities

- Undertake genetic sampling to resolve taxonomy, especially in areas where there is contact between the two greater glider species.
- Improve understanding of actions that can be undertaken to improve rates of survival and recovery in climate-affected populations.
- Identify areas likely to be climate refuges for greater glider under robust scenarios of climate change.
- Improve understanding of actions that can be undertaken to improve rates of survival and recovery following major bushfires (including characteristics of refuges, role of patchiness in fire severity, and interactions with habitat quality and disturbance history).
- Support the development of guidelines for fire management by assessing the impacts of fire management and different fire regimes (including frequency and intensity) on habitat, subpopulation size and hollow availability.
- Define appropriate levels of timber harvesting exclusion, and hollow-bearing tree retention and recruitment, to maintain population size and persistence across the species' distribution. Assess and monitor the species' response to current timber harvesting prescriptions and revise as required, noting that the effectiveness of prescriptions may differ on a regional basis depending on forest type.
- To support protection and restoration activities, improve understanding of the species' behaviours, and landscape and habitat features, that promote or constrain genetic and functional connectivity between greater glider habitat patches.
- Investigate ways to improve the effectiveness of artificial structures for mitigation of impacts on greater gliders. Research should aim to evaluate effectiveness at a scale likely to be significant for subpopulation-level recovery rather than isolated instances of use (e.g. genetic connectivity provided by glide poles over transport routes, feasibility of artificial hollows and nest boxes to sustain populations).
- Improve understanding of the species' diet and life history, especially in areas where populations have declined. Determine the likely effects of increased temperatures and drought on food supply, behaviour and survival.
- Identify priority isolated subpopulations for conservation (for example Gregory Range/Gilbert Plateau).

Recovery plan

The Committee recommends that there should be a recovery plan for *Petauroides minor* (greater glider (northern)). Stopping decline and supporting recovery is complex, due to the requirement for a high level of planning to abate the threats, knowledge gaps relating to addressing climate change as a key threat, a highly adaptive management process and a high level of support by key stakeholders. Existing mechanisms are not adequate to address these needs.

Links to relevant implementation documents

[Threat abatement plan for predation by feral cats 2015]

Conservation Advice and Listing Assessment references

References cited in the advice

Beale PK, Marsh KJ, Foley WJ & Moore BD (2018) A hot lunch for herbivores: physiological effects of elevated temperatures on mammalian feeding ecology. *Biol Rev Camb Philos Soc* 93, 674–92.

Berry LE, Driscoll DA, Banks SC & Lindenmayer DB (2015) The use of topographic fire refuges by the greater glider (*Petauroides volans*) and the mountain brushtail possum (*Trichosurus cunninghami*) following a landscape-scale fire. *Australian Mammalogy* 37, 39–45.

Collett R (1887) On a collection of mammals from Central and Northern Queensland. *Zoologische Jahrbuecher* 2, 829–940.

Coman BJ & Brunner H (1972) Food habits of the feral house cat in Victoria. *Journal of Wildlife Management* 86, 848–853.

Comport SS, Ward SJ & Foley WJ (1996) Home ranges, time budgets and food tree use in a high density tropical population of greater gliders, *Petauroides volans minor* (Pseudocheiridae: Marsupialia). *Wildlife Research* 23, 401–419.

CSIRO (Commonwealth Scientific & Industrial Research Organisation) (2020) *State of the Climate 2020*. Available on the Internet at: https://www.csiro.au/en/Showcase/state-of-the-climate

DAWE (Department of Agriculture, Water and the Environment) (2021) Area of occupancy and extent of occurrence for *Petauroides minor*. Unpublished report. Department of Agriculture, Water and the Environment (Commonwealth), Canberra.

DeGabriel JL, Moore BD, Foley WJ & Johnson CN (2009) The effects of plant defensive chemistry on nutrient availability predict reproductive success in a mammal. *Ecology* 90, 711–719.

Eyre TJ (2002) *Habitat preferences and management of large gliding possums in southern Queensland.* Ph.D. thesis, Southern Cross University, Lismore.

Eyre TJ (2004) Distribution and conservation status of the possums and gliders of southern Queensland, in RL Goldingay & SM Jackson (eds) *The Biology of Australian Possums and Gliders*. Surrey Beatty and Sons Chipping Norton, New South Wales. pp. 1-25.

Eyre TJ (2006) Regional habitat selection by large gliding possums at forest stand and landscape scales in southern Queensland, Australia. I. Greater glider (*Petauroides volans*). *Forest Ecology and Management* 235, 270–282.

Eyre TJ, Smith GC, Venz MF, Mathieson MT, Hogan LD & Starr C (2021) *Species specific advice: Greater gliders, Queensland*. Queensland Herbarium, Department of Environment and Science, Queensland Government. Brisbane.

Foley WJ, Kehl JC, Nagy KA, Kaplan IR & Boorsboom AC (1990) Energy and water metabolism in free-living greater gliders *Petauroides volans*. *Australian Journal of Zoology* 38, 1–10.

Gleadow RM, Foley WJ & Woodrow IE (1998) Enhanced CO_2 alters the relationship between photosynthesis and defense in cyanogenic *Eucalyptus cladocalyx*. *Plant, Cell and Environment* 21, 12–22.

Goldingay RL (2012) Characteristics of tree hollows used by Australian arboreal and scansorial mammals. *Australian Journal of Zoology* 59, 277–294.

Hancock N, Harris R, Broadhurst L & Hughes L (2018) *Climate-ready revegetation. A guide for natural resource managers.* Version 2. Sydney, Macquarie University. Available on the internet at: https://www.mq.edu.au/__data/assets/pdf_file/0006/807666/Climate-Reveg-Guide-v2-2018.pdf

Harrington GN & Sanderson KD (1994) Recent contraction of wet sclerophyll forest in the wet tropics of Queensland due to invasion by rainforest. *Pacific Conservation Biology* 1, 319–327.

Henry SR (1984) Social organisation of the greater glider (*Petauroides volans*) in Victoria, in AP Smith & ID Hume (eds) *Possums and Gliders.* Surrey Beatty and Sons, Chipping Norton. pp. 221–228.

Jackson S & Groves C (2015) *Taxonomy of Australian mammals*. CSIRO Publishing, Clayton South.

Jones E & Coman BJ (1981) Ecology of the Feral Cat, Felis *catus* (L.), in South-Eastern Australia I. Diet. *Wildlife Research 8*, 537–547.

Legge S, Woinarski JCZ, Garnett ST, Geyle H, Lintermans M, Nimmo DG, Rumpff L, Scheele BC, Southwell DG, Ward M, Whiterod NS, Ahyong ST, Blackmore CJ, Bower DS, Brizuela-Torres D, Burbidge AH, Burns PA, Butler G, Catullo R, Dickman CR, Doyle K, Ensbey M, Ehmke G, Ferris J, Fisher D, Gallagher R, Gillespie GR, Greenlees MJ, Hayward-Brown B, Hohnen R, Hoskin CJ, Hunter D, Jolly C, Kennard M, King A, Kuchinke D, Law B, Lawler I, Lawler S, Loyn R, Lunney D, Lyon J, MacHunter J, Mahony M, Mahony S, McCormack RB, Melville J, Menkhorst P, Michael D, Mitchell N, Mulder E, Newell D, Pearce L, Raadik TA, Rowley J, Sitters H, Spencer R, Valavi R, Ward M, West M, Wilkinson DP & Zukowski S (2021) *Estimates of the impacts of the 2019-2020 fires on populations of native animal species*. NESP Threatened Species Recovery Hub project 8.3.2 report. Brisbane, Australia.

Kavanagh RP (2000) Effects of variable-intensity logging and the influence of habitat variables on the distribution of the greater glider *Petauroides volans* in montane forest, southeastern New South Wales. *Pacific Conservation Biology 6*, 18–30.

Kavanagh RP (2004) Distribution and conservation status of possums and gliders in New South Wales, in RL Goldingay & SM Jackson (eds) *The Biology of Australian Possums and Gliders*. Surrey Beatty and Sons, Chipping Norton. pp. 130–148.

Kavanagh RP & Bamkin KL (1995) Distribution of nocturnal forest birds and mammals in relation to the logging mosaic in south-eastern New South Wales, Australia. *Biological Conservation* 71, 41–53.

Kavanagh RP & Webb GA (1998) Effects of variable-intensity logging on mammals, reptiles and amphibians at Waratah Creek, south-eastern New South Wales. *Pacific Conservation Biology* 4, 326–347.

Kavanagh RP & Wheeler RJ (2004) Home range of the greater glider *Petauroides volans* in tall montane forest of southeastern New South Wales, and changes following logging. In RL Goldingay & SM Jackson (eds) *The Biology of Australian Possums and Gliders*. Surrey Beatty and Sons, Sydney. pp. 413–425.

Kearney MR, Wintle BA & Porter WP (2010) Correlative and mechanistic models of species distribution provide congruent forecasts under climate change. *Conservation Letters* 3, 203–213.

Kehl J & Borsboom A (1984) Home range, den tree use and activity patterns in the greater glider (*Petauroides volans*), in AP Smith ID Hume (eds) *Possums and Gliders*. Surrey Beatty and Sons, Chipping Norton. pp. 229–236.

Lawler IR, Foley WJ, Woodrow IE & Cork SJ (1997) The effects of elevated CO_2 atmospheres on the nutritional quality of Eucalyptus foliage and its interaction with soil nutrient and light availability. *Oecologia* 109, 59–68.

Lindenmayer DB, Lacy RC & Pope ML (2000) Testing a simulation model for population viability analysis. *Ecological Applications* 10, 580–597.

Lindenmayer DB, Blanchard W, McBurney L, Blair D, Driscoll D, Smith AL & Gill AM (2013) Fire severity and landscape context effects on arboreal marsupials. *Biological Conservation* 167, 137–148.

McCarthy MA & Lindenmayer DB (1999a) Conservation of the greater glider (*Petauroides volans*) in remnant native vegetation within exotic plantation forest. *Animal Conservation* 2, 203–209.

McCarthy MA & Lindenmayer DB (1999b) Incorporating metapopulations dynamics of greater gliders into reserve design in disturbed landscapes. *Ecology* 80, 651–667.

McGregor DC, Padovan A, Georges A, Krockenberger A, Yoon H & Youngentob KN (2020) Genetic evidence supports three previously described species of greater glider, *Petauroides volans, P. minor, and P. armillatus. Scientific Reports, Nature Conservation* 10, 19284.

McKiernan AB, Hovenden MJ, Brodribb TJ, Potts BM, Davies NW & O'Reilly-Wapstra JM (2014) Effect of limited water availability on foliar plant secondary metabolites of two Eucalyptus species. *Environmental and Experimental Botany* 105, 55-64.

McInnes K et al. (2015) Wet Tropics Cluster Report, Climate Change in Australia Projections for Australia's Natural Resource Management Regions: Cluster Reports, eds. Ekström, M. et al., CSIRO and Bureau of Meteorology, Australia.

McLean CM, Kavanagh RP, Penman T & Bradstock R (2018) The threatened status of the hollow dependent arboreal marsupial, the greater glider (*Petauroides volans*), can be explained by impacts from wildfire and selective logging. *Forest Ecology and Management* 415-416, 19–25.

McKay GM (1989) Petauridae, in DW Walton & BJ Richardson (eds) *Fauna of Australia. Vol. 1B. Mammalia.* Australian Government Printing Service. Canberra. pp. 665–679.

McKay GM (2008) Greater glider *Petauroides volans*, in S Van Dyck & R Strahan (eds) *The Mammals of Australia*. Third edition. Reed New Holland, Sydney. pp. 240–242.

Moise A. et al. (2015) Monsoonal North Cluster Report, Climate Change in Australia Projections for Australia's Natural Resource Management Regions: Cluster Reports, eds. Ekström, M. et al., CSIRO and Bureau of Meteorology, Australia.

Moore BD, Wallis IR, Marsh KJ & Foley WJ (2004) The role of nutrition in the conservation of the marsupial folivores of eucalypt forests, in D Lunney (ed) *Conservation of Australia's Forest Fauna*. Second edition. pp 549–575.

Pope ML, Lindenmayer DB & Cunningham RB (2004) Patch use by the greater glider (*Petauroides volans*) in a fragmented forest ecosystem. I. Home range size and movements. *Wildlife Research* 31, 559–568.

Rübsamen K, Hume ID, Foley WJ & Rübsamen U (1984) Implications of the large surface area to body mass ratio on the heat balance of the greater glider *Petauroides volans*. *Journal of Comparative Physiology, B-Biochemical, Systemic, and Environmental Physiology* 154, 105–111.

Smith GC, Mathieson M & Hogan L (2007) Home range and habitat use of a low-density population of greater glider, *Petauroides volans* (Pseudocheiridae: Marsupialia), in a hollow-limiting environment. *Wildlife Research* 34, 472–483.

Smith P & Smith J (2018) Decline of the greater glider (*Petauroides volans*) in the lower Blue Mountains, New South Wales. *Australian Journal of Zoology* 66, 103–114.

Smith P & Smith J (2020) Future of the greater glider (*Petauroides volans*) in the Blue Mountains, New South Wales. *Proceedings of the Linnean Society of New South Wales* 142, 55–66.

Standards Reference Group SERA (2021) *National Standards for the Practice of Ecological Restoration in Australia*. Edition 2.2. Society for Ecological Restoration Australasia. Available on the internet at: http://www.seraustralasia.com/standards/home.html

Starr CR, Hughes RT, Hemmings MF, Coase JF & Jess MD (2021) Field studies of a high elevation population of northern greater glider *Petauroides volans minor* in the Bluff State Forest, Far North Queensland. *Australian Zoologist*. Available on the internet at: https://doi.org/10.7882/AZ.2021.003

Taylor BD & Goldingay RL (2009) Can road-crossing structures improve population viability of an urban gliding mammal? *Ecology and Society* 14, 13.

Taylor AC, Tyndale-Biscoe CH & Lindenmayer DB (2007) Unexpected persistence on habitat islands: genetic signatures reveal dispersal of a eucalypt-dependent marsupial through a hostile pine matrix. *Molecular Ecology 16*, 2655–2666.

Tyndale-Biscoe CH & Smith RFC (1969a) Studies on the marsupial glider *Schoinobates volans* (Kerr). III. Response to habitat destruction. *Journal of Animal Ecology* 38, 651–659.

Tyndale-Biscoe CH & Smith RFC (1969b) Studies on the marsupial glider, *Schoinobates volans* (Kerr). II. Population structure and regulatory mechanisms. *Journal of Animal Ecology* 38, 637–650.

Vanderduys EP, Kutt AS & Kemp JE (2012) Upland savannas: the vertebrate fauna of largely unknown but significant habitat in north-eastern Queensland. *Australian Zoologist* 36, 59–74.

van der Ree R (1999) Barbed wire fencing as a hazard for wildlife. *The Victorian Naturalist* 116, 210–217.

van der Ree R, Ward SJ & Handasyde KA (2004) Distribution and conservation status of possums and gliders in Victoria, in RL Goldingay & SM Jackson (eds) *The Biology of Australian Possums and Gliders*. Surrey Beatty and Sons, Sydney. pp. 91–110.

van Dyck S & Strahan R (2008) *The Mammals of Australia*. Reed New Holland, Sydney.

Wagner B, Baker PJ, Stewart SB, Lumsden LF, Nelson JL, Cripps JK, Durkin LK, Scrogge M & Nitschke CR (2020) Climate change drives habitat contraction of a nocturnal arboreal marsupial at its physiological limits. *Ecosphere* 11, e03262. Available on the internet at: https://doi.org/10.1002/ecs2.3262

Winter JW, Dillewaard HA, Williams SE & Bolitho EE (2004) Possums and gliders of north Queensland: distribution and conservation status, in RL Goldingay & SM Jackson (eds) *The Biology of Australian Possums and Gliders*. Surrey Beatty and Sons, Sydney. pp. 26–50.

Woinarski JCZ, Burbidge AA & Harrison PL (2014) *The Action Plan for Australian Mammals 2012*. CSIRO Publishing, Collingwood.

Youngentob KN, Wallis IR, Lindenmayer DB, Wood JT, Pope ML & Foley WJ (2011) Foliage influences tree choice and landscape use of a gliding marsupial folivore. *Journal of Chemical Ecology* 37, 71–84.

Youngentob KN, Lindenmayer DB, Marsh KJ, Krockenberger AK & Foley WJ (2021) Food intake: an overlooked driver of climate change casualties? *Trends Ecol Evol*, 36, 676–678.

Other sources cited in the advice

Armstrong KN (2021) Personal communication via email, 24 June 2021. University of Adelaide.

Jones KE, Bielby J, Cardillo M, Fritz SA, O'Dell J, Orme CDL, Safi K, Sechrest W, Boakes EH, Carbone C, Connolly C, Cutts MJ, Foster JK, Grenyer R, Habib M, Plaster CA, Price SA, Rigby EA, Rist J, Teacher A, Bininda-Emonds ORP, Gittleman JL, Mace GM & Purvis A (2009) PanTHERIA: a species-level database of life history, ecology, and geography of extant and recently extinct mammals. *Ecology* 90, 2648.

Pacifici M, Santini L, Di Marco M, Baisero D, Francucci L, Grottolo Marasini G, Visconti P, Rondinini C (2013) Database on generation length of mammals. 5427 data records. Online at http://doi.org/10.5061/dryad.gd0m3, version 1.0 (last updated on 2013-08-27), Resource ID: 10.5061/dryad.gd0m3, Data Paper ID: doi: 10.3897/natureconservation.5.5734

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

Attachment A: Listing Assessment for *Petauroides minor* (greater glider (northern))

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	unknown	>10 000	> 30 000	The maximum plausible value of >30 000 individuals was estimated by Woinarski et al. (2014) for the northern subspecies *P. v. minor* as part of the Action Plan for Australian Mammals. Although this estimate has low reliability, it is unlikely that there are fewer than 10 000 mature individuals.
Trend	contracting			
Generation time (years)	7	6	8	The greater glider can live for 15 years (Jones et al. 2009) and reaches sexual maturity at two years of age (Tyndale-Biscoe & Smith 1969b), suggesting a generation length of six to eight years (Pacifici et al. 2013; Woinarski et al. 2014).
Extent of occurrence	48 946 km^2	43 655 km^2	48 946 km^2	Woinarski et al. (2014) estimated the extent of occurrence (EOO) as 43 655 km^2, calculated using records from 1993 to 2012. The 48 946 km^2 figure was based on the mapping of point records from 1997 to 2017, obtained from state governments, museums and CSIRO (DAWE 2021). The EOO was calculated using a minimum convex hull, based on the IUCN Red List Guidelines 2019.

Metric	**Estimate used in the assessment**	**Minimum plausible value**	**Maximum plausible value**	**Justification**
Trend	contracting			Woinarski et al. (2014) considered that the broad extent of occurrence (EOO) is unlikely to have changed appreciably since European settlement. However, the species is predicted to undergo a severe range contraction due to increased temperatures as a result of climate change (Kearney et al. 2010).
Area of Occupancy	524 km^2	500 km^2	<2000 km^2	The 524 km^2 figure is based on the mapping of point records over 1997–2017, obtained from state governments, museums and CSIRO (DAWE 2021). The AOO was calculated using a 2x2 km grid cell method, based on the IUCN Red List Guidelines 2019. The AOO is likely to be underestimated due to limited sampling across the species range (Woinarski et al. 2014). It is not possible to determine an upper plausible estimate to adjust for the under sampling, but given the expected range contraction due to climate change (Kearney et al. 2010), the maximum plausible value is likely to be <2000 km^2.
Trend	contracting			The AOO has declined since European settlement, with loss of habitat from land clearing, fragmentation, timber harvesting, inappropriate fire regimes and climate change (Woinarski et al. (2014). The species is predicted to undergo a severe range contraction due to increased temperatures as a result of climate change (Kearney et al. 2010).
Number of subpopulations	Unknown	Unknown	>10	There is no reliable estimate for the number of subpopulations due to limited sampling across the species' range. However, Woinarski et al. (2014) estimated it as >10.
Trend	contracting			As the number of greater gliders (northern) and its AOO are continuing to decline, the number of subpopulations is also likely to be declining.
Basis of assessment of subpopulation number	The number of subpopulations is unknown as there is a limited sampling and survey effort across its range.			

Metric	Estimate used in the assessment	Minimum plausible value	Maximum plausible value	Justification
No. locations	1	1	>10	There is no robust estimate for the number of locations. However, Woinarski et al. (2014) estimated it as >10. The biophysical modelling of Kearney et al. (2010) predicted a severe range contraction for the greater glider in the Wet Tropics (approximately the range of greater glider (northern)) if there is a 3°C temperature increase. As this potentially impacts the entire population, a single location is given as the minimum value.
Trend	stable			Climate change is likely to result in a decline in the occupied range of the greater glider (northern) (Kearney et al. 2010). However, as the number of locations used in the assessment is 1, this cannot have a declining trend.
Basis of assessment of location number	Kearney et al. (2010) utilising biophysical modelling, predicted a severe and dire range contraction (~90%) for the greater glider (northern) with a 3 °C temperature increase.			
Fragmentation	No data to suggest distribution is severely fragmented.			
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 A2c+3c+4c for listing as Vulnerable

The greater glider (northern) has a generation length of six to eight years (see Table 3). In this assessment a generation length of seven years is used, which gives a timeframe of 21 years for this criterion.

In 2016, *Petauroides volans* (understood at the time to be a single species consisting of both the greater glider (northern) and greater glider (southern and central)) was assessed by the Threatened Species Scientific Committee to be eligible for listing as Vulnerable under Criterion 1 (TSSC 2016). However, the current assessment addresses the greater glider (northern) specifically, which is at the northern end of the range of the formerly assessed species. The 2016 assessment used a preponderance of data from the southern part of the range and thus fewer direct data are available for this assessment. The 2016 assessment of *P. volans* provides only a broad indication of the likely status of the greater glider (northern) in isolation. There are no robust estimates of population size or population trends of the greater glider (northern) across its distribution. In fact, there are few published studies on the species abundance at all, save for Comport et al. (1994), Vanderduys et al. (2012) and a recent study (Starr et al. 2021).

Legge et al. (2021) gave estimates of population decline separate to those caused by the 2019-20 bushfires (as well as caused by the fires). This gave an estimated overall decline for the greater glider (northern) of 18 percent (range 7-33%) over the next three generations, assuming current management conditions. This estimate does not include the impacts of future fire and droughts.

The biophysical modelling of Kearney et al. (2010) predicted a severe range contraction for the greater glider in the Wet Tropics (approximately the range of greater glider (northern)), with a 3°C temperature increase. Two correlative models (Maxent and Bioclim) predicted 98.4 percent and 94 percent range contractions respectively, while a mechanistic model (Niche Mapper) predicted a 76.3 percent decline. The CSIRO (2020) State of the Climate Report shows that the Australian average temperature has increased by up to 0.5°C since the date of the distributional data used in Kearney et al. (2010) was collected, and is projected to increase by a similar amount by 2025 (i.e. by approximately 1°C over ~ three generations). A similar temperature rise is expected over the period from the present (2021) to 2040 (approximately three generations into the future).

Approximately one third of the temperature increase modelled by Kearney et al. (2010) will occur, or has occurred, over any three generation time period relevant to this assessment. The predictions of Kearney et al. (2010) provide an endpoint, but do not describe the pattern of likely decline and thus it is not possible to precisely predict levels of range contraction. Nevertheless, they are sufficient to infer that the distribution of the greater glider (northern) both has contracted, and is projected to contract, by at least 30 percent within time periods relevant to this assessment.

The Committee considers that the species has undergone a substantial reduction in numbers over three generations (21 years for this assessment) and projected to reduce further in the future (2021–2040), equivalent to at least 30 percent, and the reduction has not ceased and the cause has not ceased. Therefore, the species 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 subpopulations; (v) number of mature individuals			
(c)	Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations; (iv) number of mature individuals			

Criterion 2 evidence

Eligible under Criterion 2 B2ab(i,ii,iii,iv,v) for listing as Vulnerable

The extent of occurrence (EOO) is estimated at 48 946 km^2, and the area of occupancy (AOO) estimated at 524 km^2. These figures are based on the mapping of point records over 1997–2017, obtained from state governments, museums and CSIRO (DAWE 2021). 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 2019. Woinarski et al. (2014) noted that the AOO, which they estimated to be 716 km^2, is likely to be a significant underestimate due to limited sampling across the occupied range. For the purposes of this assessment, the AOO is considered to be >500 km^2. It is not possible to determine an upper plausible estimate to adjust for the under sampling, but given the expected range contraction due to climate change (Kearney et al. 2010) it is judged here to be <2000 km^2.

Woinarski et al. (2014) estimated the number of locations to be >10. However, with a severe range contraction of up to 98 percent predicted due to temperature increase across the greater glider's (northern) range (Kearney et al. 2010), there can be considered to be a single location in the context of this criterion, which meets subcriterion (a). The are insufficient data to determine whether the distribution of the greater glider (northern) is severely fragmented.

The area of suitable habitat, and therefore AOO and EOO, of the greater glider (northern) are projected to continue to decline under future climate change scenarios (Kearney et al. 2010). As a consequence, the number of subpopulations and mature individuals (see also Criterion 1) are also inferred to be declining. This meets subcriterion (b)(i,ii,iii,iv,v).

After assessment of the data, the Committee considers that the AOO is limited, there is only one location, and there is a continuing decline in EOO, AOO, habitat, number of subpopulations and number of mature individuals. Therefore, the species has met the relevant elements of Criterion 2 to make it eligible for listing as Vulnerable.

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 subpopulation	≤ 50	≤ 250	≤ 1,000
	(ii) % of mature individuals in one subpopulation =	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. Winter et al. (2004) considered that the greater glider (northern) had a 'presumed large population' and was 'locally common'. Density estimates in north-eastern Qld range from 2.6 to 5.8 individuals per hectare (Comport et al. 1996; Vanderduys et al. 2012). A recent density estimates for the greater glider (northern) from The Bluff State forest range from 0.24 to 0.38 individuals per hectare in wet and dry sclerophyll forest respectively (Starr et al. 2021). These estimates suggest >10 000 individuals if applied across the species' area of occupancy (e.g. using an AOO of 500 km^2 and density estimate of 3/ha gives 16 667 individuals).

Woinarski et al. (2014) estimated the number of mature individuals to be greater than 30 000, but noted that this estimate has low reliability. With none of the greater glider (northern) habitat affected by the 2019–2020 bushfires, it is unlikely that the population of greater glider (northern) has been reduced to substantially below 30 000 mature individuals.

Following assessment of the data the Committee considers that the species is not eligible for listing in any category under this criterion as the number of mature individuals is unlikely 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 km² 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

The number of mature individuals is likely to be greater than 10 000 (see Criterion 3) and highly unlikely to be less than 1000. Additionally, the greater glider (northern) does not meet the quantitative threshold for Vulnerable under sub-criterion D2. Although the species is considered to occur at a single location, the plausible future threat (climate change) is unlikely to drive the species to critically endangered within a very short time (one generation) (see Criterion 1).

Following assessment of the data the Committee considers that the species is not eligible for listing in any category under this criterion as the number of mature individuals is not low.

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

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 minor* in the list in the Vulnerable category.

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

Cataloguing data

This publication (and any material sourced from it) should be attributed as: Department of Climate Change, Energy, the Environment and Water 2022, *Conservation advice for* Petauroides minor *(greater glider (northern))*, Canberra.

This publication is available at the SPRAT profile for *Petauroides minor* (greater glider (northern)).

Department of Climate Change, Energy, the Environment and Water
GPO Box 858, Canberra ACT 2601
Telephone 1800 900 090
Web dcceew.gov.au

Version history table

Document type	Title	Date
Conservation Advice (including listing assessment)	Conservation Advice for *Petauroides volans* (greater glider (northern))	Approved 05/07/2022
Conservation Advice (including listing assessment)	Conservation Advice for *Petauroides volans* (greater glider)	Approved 25/05/2016

Australian Government

Department of Climate Change, Energy, the Environment and Water

Conservation Advice for *Petauroides volans* (greater glider (southern and central))

In effect under the *Environment Protection and Biodiversity Conservation Act 1999* from 5 July 2022.

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

Petauroides volans © Copyright, Tyrie Starrs (from Tallaganda NSW)

Conservation status

Petauroides volans (greater glider) 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 5 May 2016.

This assessment recognises that *P. volans*, as understood in 2016 is now considered to be at least two separate species: *P. volans* (greater glider (southern and central)) and *P. minor* (greater glider (northern)) (McGregor et al. 2020).

Petauroides volans (southern and central) was assessed by the Threatened Species Scientific Committee to be eligible for listing as Endangered under Criterion 1. The Committee's assessment is at Attachment A. The Committee assessment of the species' eligibility against each of the listing criteria is:

- Criterion 1: A2abc+4bc: Endangered
- Criterion 2: Not eligible
- Criterion 3: Not eligible
- Criterion 4: Not eligible
- Criterion 5: Insufficient data

The main factors that make the species eligible for listing in the Endangered category are an overall rate of population decline exceeding 50 percent over a 21-year (three generation) period, including population reduction and habitat destruction following the 2019–20 bushfires.

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.

The current listing status of this species under relevant state or territory legislation is:

- Victoria (Vic): Vulnerable under the *Flora and Fauna Guarantee Act 1988* since June 2017,
- Australian Capital Territory (ACT): Vulnerable under the *Nature Conservation Act 2014* since May 2019,
- New South Wales (NSW): Three subpopulations listed as Endangered Populations under the *Biodiversity Conservation Act 2016* (Euroballa Local Government Area since 2007, Mount Gibraltar Reserve since 2015 and Seven Mile Beach National Park since 2016), and
- Queensland (Qld): Vulnerable under the *Nature Conservation Act 1992* (includes both the greater glider (southern and central) and greater glider (northern)) since October 2014.

Species information

Taxonomy

Conventionally accepted as *Petauroides volans* Kerr (1792).

This was formerly the only species in the genus. Two subspecies were recognised: *P. v. minor* (in north-eastern Qld) and *P. v. volans* (in south-eastern Australia) (van Dyck & Strahan 2008).

Jackson & Groves (2015) split the species into three separate species: *P. minor* (Atherton Tablelands and coastal central and northern Qld), *P. armillatus* (inland central Qld), and *P. volans* (from south-east Qld to Vic). McGregor et al. (2020) agreed with this taxonomic arrangement within *Petauroides* on the basis of genomic-scale nuclear markers and external morphological data.

A new dataset that combined the genetic resources of McGregor et al. (2020) and that of B Arbogast & K Armstrong et al. (manuscript in prep.), which included more extensive sampling throughout the range of *Petauroides* for genomic-scale markers, a mitochondrial marker dataset and cranial measurements, has supported the separate recognition of *P. minor* (KN Armstrong pers comm 24 June 2021). The dataset also provides evidence that all *Petauroides* south of the Burdekin gap (from around Proserpine) should be considered as two separate taxa, at least at the level of subspecies, with a point of contact between them in the vicinity of Coffs Harbour (KN Armstrong pers comm 24 June 2021). These two taxa need redescription, and might be elevated to the species level in the future. Until this ambiguity is resolved and the taxonomic split of *P. volans* is formally recognised by the Australian Faunal Directory, the listed entity in this Conservation Advice will be referred to as *Petauroides volans* (greater glider (southern and central)). The common name greater glider will refer to the genus *Petauroides.*

Description

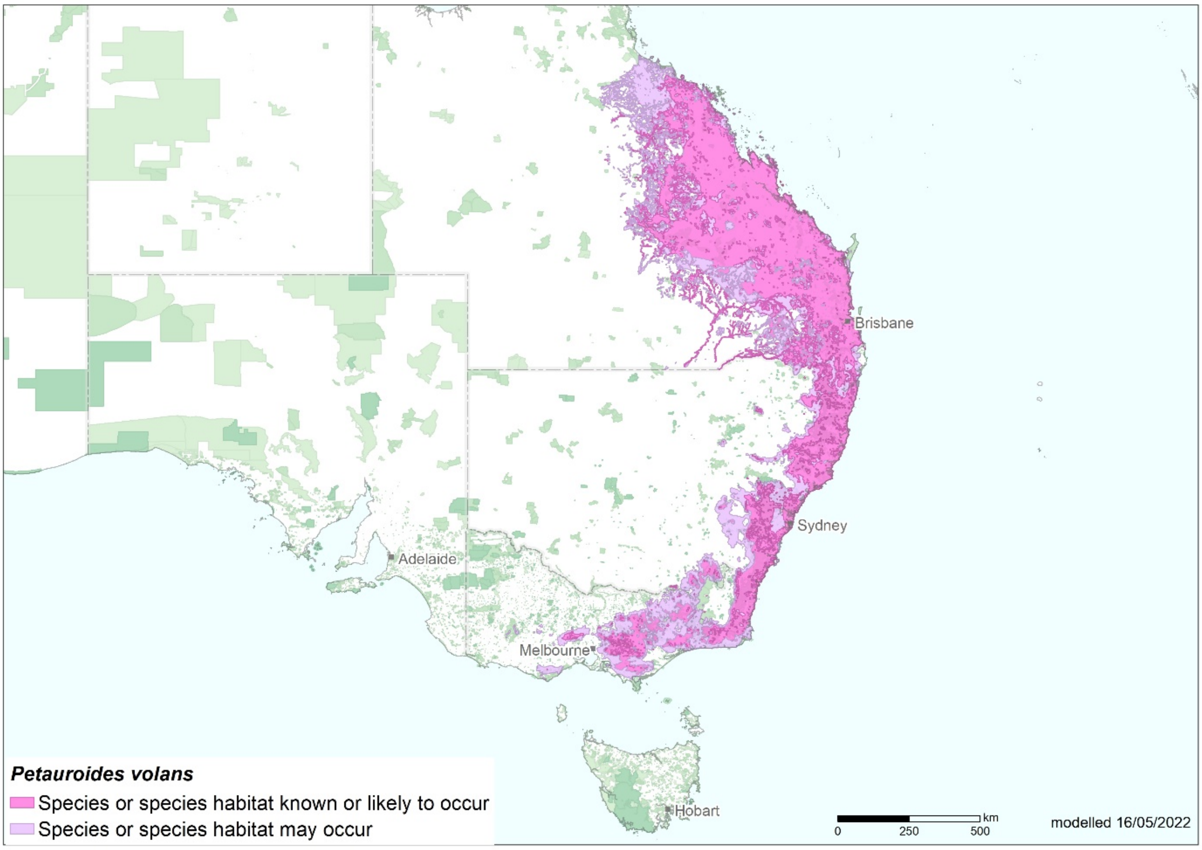
The greater glider (southern and central) is the largest gliding possum in eastern Australia. It has a head and body length of 35−46 cm, tail length of 45−60 cm, and a weight range of 900−1700 g, with females being larger than males (McKay 1989, 2008; McGregor et al. 2020). The greater glider (southern and central) has thick fur that increases its apparent size. Its fur colour is white or cream below and varies from dark grey, dusky brown through to light mottled grey and cream above. It has a long furry tail, large furry ears and a short snout. Its tail is not prehensile, and the gliding membrane extends from the forearm to the tibia (Mckay 1989, 2008).

Distribution

The greater glider (southern and central) occurs in eastern Australia, where it has a broad distribution from around Proserpine in Qld, south through NSW and the ACT, to Wombat State Forest in central Vic (McGregor et al. 2020; B Arbogast & KN Armstrong et al. unpublished data; OZCAM records: Atlas of Living Australia 2021). It occurs across an elevational range of 0−1200 m above sea level (a.s.l) (Kavanagh 2004). The distribution appears to be restricted in the ACT, where the species is only known from the Lower Cotter Catchment and Namadgi National Park (Canberra Nature Map 2019). The species formerly occurred in Booderee National Park but appears to have been extirpated from that location in the mid-late 2000s.

The species' distribution overlaps with some World Heritage Areas, including the Gondwana Rainforests of Australia and the Blue Mountains. It also occurs on some Commonwealth lands, including the Shoalwater Bay Training Area (managed by the Department of Defence) near Rockhampton (Queensland Herbarium 2018).

The extent of occurrence (EOO) is unlikely to have changed appreciably since European settlement (Eyre 2004; Kavanagh 2004; van der Ree et al. 2004). However, the area of occupancy (AOO) has decreased substantially, mostly due to land clearing. This area is probably continuing to decline due to further clearing, fragmentation impacts, edge effects, bushfire, climate change and some forestry activities (Eyre 2005; Lindenmayer et al. 2011; Youngentob et al. 2012; Berry et al. 2015; McLean et al. 2018; Wagner et al. 2020). In addition, some subpopulations in undisturbed, intact habitat have disappeared or undergone rapid decline (Lindenmayer et al. 2011, 2018b; Smith & Smith 2018). The species appears to have been extirpated from Booderee National Park, where it has not been recorded since 2006, for reasons that are unclear (Lindenmayer 2018b). The steep decline of subpopulations in the Blue Mountains World Heritage Area is likely to be due to increased temperatures as a result of climate change (Smith & Smith 2018; Wagner et al. 2020). The existence of these recent declines suggests that many unmonitored subpopulations of the greater glider (southern and central) are likely similarly declining.



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 greater glider (southern and central) is poorly known. However, the habitats and area in which it is found have a long and profound history of management by Indigenous Australians. Stacie Nicho Piper, Wurundjeri Traditional Owner, states that: "All native animals on Country are our totems, spirit protectors, including the greater glider. They hold significant roles in the balance of country and our spiritual connections/values. When they are affected, country is affected, we as people are affected."

Relevant biology and ecology

General habitat

The greater glider (southern and central) is an arboreal nocturnal marsupial, predominantly solitary and largely restricted to eucalypt forests and woodlands of eastern Australia. It is typically found in highest abundance in taller, montane, moist eucalypt forests on fertile soils, with relatively old trees and abundant hollows – e.g. north-eastern NSW (Andrews et al. 1994; Smith et al. 1994a,b), south-eastern NSW (Kavanagh 2000), eastern Vic (van der Ree et al. 2004) – but also occurs in drier habitats in south-eastern Qld (Eyre 2004). The distribution may be patchy even in continuous areas of habitat, such as Tantawangalo State Forest in NSW (Kavanagh 2000). It is likely that only a proportion of forest in potential habitat areas is suitable for the species, as the structural attributes of the forest overstorey and forage quality it relies on vary considerably across the landscape (Eyre 2002; Youngentob et al. 2011).

Den trees

During the day the greater glider (southern and central) shelters in tree hollows, with a particular preference for large hollows (diameter >10 cm) in large, old trees (Henry 1984; Kehl & Borsboom 1984; Lindenmayer et al. 1991; Smith et al. 2007; Goldingay 2012). Both live and standing dead trees are used for denning (Goldingay 2012), however the species prefers to use live hollow-bearing trees when adequate numbers are available (Kehl & Borsboom 1984; Kavanagh & Wheeler 2004; Lindenmayer et al. 2004). Multiple dens are used by an individual. Near Tumut in NSW, individuals used a few den trees frequently, located near core home-range areas, and numerous others infrequently (Lindenmayer et al. 2004). In south-eastern Queensland, 4−20 different den trees were used by individuals (Smith et al. 2007).

The probability of occurrence of the species is positively correlated with the availability of tree hollows (Andrews et al. 1994; Smith et al. 1994a,b; Lindenmayer et al. 2020), which is a key limiting resource. Greater gliders (southern and central) can be found in regrowth forest provided sufficient hollows are present (Macfarlane 1988; Lindenmayer et al. 1990a), and conversely are absent when there are insufficient hollows. In the Grafton/Casino region of NSW, the species was not recorded from surveyed sites containing fewer than six tree hollows per hectare (Smith et al. 1994). In southern Qld, the species appears to require at least 2–4 live den trees for every 2 ha of suitable forest habitat (Eyre 2002). Most hollow-bearing trees used for denning by arboreal and scansorial mammals are at least 100 years of age (Mackowski 1984; Wormington & Lamb 1999; Gibbons & Lindenmayer 2002; Goldingay 2012). However, the size and age at which suitable hollows develop depends on tree species and climate.

Some tree species form hollows more readily than others (Gibbons & Lindenmayer 2002), and the greater glider appears to select these for denning. Near Tumut in NSW, the greater glider used *Eucalypts viminalis* (manna gum) and *E. dalrympleana* (mountain gum) more frequently than other species, and these species supported the highest numbers of hollows in this region (Lindenmayer et al. 2004). In south-eastern Qld the species showed a strong preference for three den-tree species (*E. acmenoides* (broad-leaved white mahogany), *E. fibrosa* (red ironbark) and *E. tereticornis* (forest red gum)) due to their availability as hollow-bearing trees (Kehl & Borsboom 1984; Smith et al. 2007). In five studies across its geographic range, the greater glider was found to utilise 25 different tree species for denning (Goldingay 2012).

Diet

The greater glider (southern and central) is primarily folivorous, with a diet mostly comprising eucalypt leaves supplemented by buds and flowers (Kehl & Borsboom 1984; Kavanagh & Lambert 1990; van der Ree et al. 2004). It feeds from a restricted range of eucalypt species, such as *E. radiata* (narrow-leaved peppermint) in Vic (Henry 1995), manna gum in south-eastern NSW (Kavanagh & Lambert 1990), and *E. moluccana* (grey box) in south-eastern Qld (Smith et al. 2007). The tree species favoured by greater gliders varies regionally. It favours forests with a diversity of eucalypt species, due to seasonal variation in growth and nutrient content of its preferred tree species (Kavanagh 1984). Approximately 85 percent of the greater glider's water requirements are provided by consumed leaves (Foley et al. 1990). Free water is presumably obtained from dew condensation on leaf surfaces (Rübsamen et al. 1984).

Life history

Females give birth to a single young from March to June (Tyndale-Biscoe & Smith 1969b; McKay 2008). Sexual maturity is reached in the second year (Tyndale-Biscoe & Smith 1969b). Longevity has been estimated at 15 years (Jones et al. 2009), and generation length is estimated to be six to eight years (Pacifici et al. 2013; Woinarski et al. 2014). The relatively low reproductive rate (Henry 1984) may render small populations in isolated remnants prone to extinction (van der Ree 2004; Pope et al. 2004).

Home ranges and densities

Home ranges are typically relatively small (1–4 ha: Henry 1984; Kehl & Borsboom 1984; Gibbons & Lindenmayer 2002; Pope et al. 2004), but are larger (up to 19 ha) in forests on less fertile sites and in more open woodlands (Smith et al. 2007). Males tend to have larger home ranges than females in the same region (Kavanagh & Wheeler 2004; Pope et al. 2004), and male home ranges tend not to overlap (Henry 1984; Kavanagh & Wheeler 2004; Pope et al. 2004).

Densities vary significantly across the greater glider's range. Average densities have been found to range from 0.6 to 2.8 individuals per hectare in Vic (Henry 1984; van der Ree et al. 2004; Lindenmayer et al. 2011; Nelson et al. 2018), 0.2 to 3.0 individuals per hectare in NSW (Tyndale-Biscoe & Smith 1969b; Kavanagh 1984, 1995; Pope et al. 2004; Lindenmayer et al. 2011; Smith & Smith 2018; Vinson et al. 2020), and 0.2 to 2.3 individuals per hectare in south-eastern Qld (Kehl & Borsboom 1984; Smith et al. 2007; Ferguson et al. 2018).

Disturbance ecology

The greater glider is particularly sensitive to forest clearance (Tyndale-Biscoe & Smith 1969a) and to intensive timber harvesting (Kavanagh & Bamkin 1995; Kavanagh & Webb 1998; Kavanagh & Wheeler 2004; Mclean et al. 2018), although responses vary according to landscape context and the extent of tree removal and retention (Kavanagh 2000; Taylor et al. 2007).

Large hollow-bearing trees are in rapid decline in some landscapes (Lindenmayer et al. 2017a,b) primarily due to timber production practices and bushfires that prevent trees growing to an age when they might produce hollows (Lunney 1987; Lindenmayer et al. 2018b). Site-level, tree-level (e.g. size, extent of decay) and landscape factors all appear to influence the rate of collapse of hollow-bearing trees. Lindenmayer et al. (2018a) found that the probability of collapse of hollow-bearing trees in remnant 1 ha patches increased with an increasing amount of logged or burned areas in the surrounding landscape (within a 2 km radius), most likely due to altered wind patterns from a reduction in forest cover. The decline in hollow-bearing trees is a concern for recovery as the greater glider is dependent on this habitat feature, and the development of hollows in suitable tree species can take over a century (Mackowski 1984). Additionally, the abundance of hollow-bearing trees may be an overestimate of the actual number that are suitable for occupation by wildlife, as only one in every 3–5 hollow-bearing trees within montane ash forests is occupied by arboreal marsupials (Lindenmayer et al. 1990b, 1993). A decline or loss of hollow-bearing trees reduces the numbers of greater gliders in the landscape (Mclean et al. 2018).

Greater gliders are sensitive to fragmentation (McCarthy & Lindenmayer 1999a,b; Lindenmayer et al. 2000; Eyre 2006; Taylor & Goldingay 2009). Although greater gliders have small home ranges, their low reproductive rate and sensitivity to disturbance means they tend to become locally extinct in small and fragmented habitat patches. Greater gliders disperse poorly across vegetation that is not native forest, and so do not readily recolonise isolated sites from which they have been lost (Pope et al. 2004). In a study of remnant patches <1 ha to >50 ha in size, Youngentob et al. (2013) found that the probability of occurrence of greater gliders increased as the area of remnant habitat increased. It is difficult to identify the smallest patch size used, as this likely varies across the range depending on vegetation type, quality, connectivity and other environmental factors. Greater gliders have been found in habitat patches <10 ha in some fragmented and remnant forest patches in the southern part of their geographic range (Pope et al. 2004; Lindenmayer 2002), but may require larger habitat patches in Queensland (Eyre 2006).

The greater glider is sensitive to bushfire (Lunney 1987; Andrews et al. 1994; Lindenmayer et al. 2011; Mclean et al. 2018) and is slow to recover following major fires (Kavanagh 2004). Substantial losses or declines of greater glider populations have been documented after fires (see Table 1), through direct mortality and indirect impacts on habitat (McLean et al. 2018).

Over the longer term, repeated disturbance such as intense or too-frequent fires degrades greater glider habitat by changing the composition, structure and nutrient profile of forests. Fire can increase or decrease the amount of tree hollows depending on the fire regime, age and species of the dominant trees, and disturbance history. Fire can destroy live and dead hollow-bearing trees, particularly in young forests because smaller diameter trees have a lower capacity to survive burning (Gibbons & Lindenmayer 2002). Fire can also result in extensive losses of dead hollow-bearing trees (Lindenmayer et al. 2012), though these are less preferred by greater gliders. Eyre et al. (2010) found that the density of such trees was substantially reduced by both low-frequency and high-intensity fires (wildfire), and by high-frequency and low-intensity burns associated with stock grazing management. Too-frequent fires can change the floristic composition and nutritional profile of glider habitat if a fire returns before the dominant trees preferred by gliders can mature and reproduce (Lindenmayer et al. 2013, Au et al. 2019). A positive feedback loop may also occur as dense regrowth is at higher risk of burning at high severity (Taylor et al. 2014).

Greater glider populations are slow to recover and recolonise burnt sites following fire and may take decades to return (Andrew et al. 2014; Lumsden et al. 2013; Vic SAC 2015; Lindenmayer et al. 2021), due to the low reproductive rate of the species and its limited dispersal capabilities. Habitat fragmentation can compound the impact of fires by hampering the recolonisation ability of greater gliders. Recovery depends on there being no further major fires in the interim (Vic SAC 2015). Major bushfires in 2003, 2006–2007 and 2009 burnt much of the species' range in Victoria, and further fragmented its distribution as evidenced by surveys and species records (Lumsden et al. 2013; Vic SAC 2015). Since the 2009 fires, spotlighting records of greater gliders (southern and central) in the Kinglake East Bushland Reserve and nearby areas have significantly declined and not yet recovered (C Cobern 2015. pers comm 9 November). Unburnt areas provide critical refuges for greater gliders in regions heavily impacted by fires, as they may be the only areas with the requisite habitat attributes within extensive landscapes for many years (Lumsden et al. 2013; Chia et al. 2015).

Habitat critical to the survival

Within the same forest type (with similar habitat structure and tree species composition), the species' occurrence is positively correlated with levels of foliar nutrients (Braithwaite et al. 1983), amount of foliage (Davey 1984), canopy productivity (Youngentob et al. 2015), stand age (Lindenmayer et al. 1990a), overstorey basal area (Kavanagh 1987; Incoll et al. 2001), tree hollow abundance (Lindenmayer et al. 1990b; Lindenmayer et al. 2013), patch size (Incoll et al. 2001; Youngentob et al. 2015) and connectivity (Youngentob et al. 2013).

Habitat critical to survival for the greater glider (southern and central) may be broadly defined as (noting that geographic areas containing habitat critical to survival needs to be defined by forest type on a regional basis):

- large contiguous areas of eucalypt forest, which contain mature hollow-bearing trees[1] and a diverse range of the species' preferred food species in a particular region; and
- smaller or fragmented habitat patches connected to larger patches of habitat, that can facilitate dispersal of the species and/or that enable recolonization; and
- cool microclimate forest/woodland areas (e.g. protected gullies, sheltered high elevation areas, coastal lowland areas, southern slopes); and
- areas identified as refuges under future climate changes scenarios; and
- short-term 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.

[1] Tree hollows can be difficult to detect in ground-based surveys. The presence of trees with basal diameter > 30 cm can be used as a proxy measure for tree hollows used by greater gliders in Queensland (Eyre et al. 2021).

Habitat meeting any one of the criteria above is considered habitat critical to the survival of greater glider (southern and central), irrespective of the current abundance or density of greater gliders or the perceived quality of the site. Forest areas currently unoccupied by the greater glider (southern and central) may still represent habitat critical to survival, if the recruitment of hollow-bearing trees as the forest ages could allow the species to colonise these areas and ensure persistence of a subpopulation.

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.

Given its Endangered status, all populations of the greater glider (southern and central) are important for the conservation of the species across its range. Due to the species' low fecundity and limited dispersal capabilities, areas where the species has become locally extinct are not readily recolonised. Coastal populations may be important for maintaining genetic diversity, as they are geographically distinct from inland populations (DoEE 2016b).

Threats

Key threats to the greater glider (southern and central) are frequent and intense bushfires, inappropriate prescribed burning, climate change, land clearing and timber harvesting (Table 1). There are synergies between these threats, and their combined impact needs to be considered in the recovery of the species. Loss and fragmentation of habitat has already occurred in many areas of the species' range (Lindenmayer et al. 2011; Youngentob et al. 2013), and the unprecedented 2019-20 bushfires have increased pressure on its remaining habitat.

Table 1 Threats impacting the greater glider (southern and central)

Threat	Status and severity [a]	Evidence
Habitat loss, disturbance and modification		
Inappropriate fire regimes	• Timing: current and future • Confidence: observed • Consequence: catastrophic • Trend: increasing • Extent: across the entire range	<u>Extensive severe bushfires</u> Substantial population losses or declines have been documented in and after high severity bushfires (Lindenmayer et al. 2013; Berry et al. 2015; McLean et al. 2018). Losses can occur as a result of direct mortality due to lethal heating or suffocation from smoke, or indirect mortality due to the loss of key habitat features and resources (McLean et al. 2018). A single fire in a ten-year period is capable of reducing the abundance of greater gliders (southern and central) by more than half (McLean et al. 2018). Declines can occur even in small fire refuges; Berry et al. (2015) found that the species was significantly less abundant in wet unburnt forest gullies within the extent of a major fire compared to similar sites outside. Occurrence at burnt sites is influenced by landscape context. Lindenmayer et al. (2020) found that the probability of occurrence of greater gliders (southern and central) is negatively associated with increasing extent of fire in the surrounding landscape. Chia et al. (2015) found that Glider abundance was lower in areas affected by high-intensity fires than in areas where fires

Threat	Status and severity [a]	Evidence
		burnt only the understorey, and that abundance increased with increasing amount of unburnt and understorey-only burnt forest within a 1 km radius. These results suggest that unburnt areas, e.g. gullies, can serve as post-fire refuges and assist recolonization of severely burned forest. Remaining unburnt areas provide critical refuges for species heavily impacted by fires, as they will be the only areas with mature habitat within extensive landscapes for many years (Dickman et al. 2020). In 2019-20, following years of drought (DPI 2020) and Australia's hottest and driest year on record in 2019, catastrophic wildfire conditions culminated in fires that covered an unusually large area of eastern and southern Australia and burnt with high severity in many places (Boer et al. 2020). The full impact of the 2019-20 bushfires has yet to be determined. However, an estimated 40% of the distribution of the greater glider (southern and central) overlapped with the areas affected by the bushfires (Legge et al. 2021). A population decline analysis for the greater glider (southern and central) that incorporates spatial variation in fire severity plus estimated declines for differing fire severity classes, provided an estimate of overall decline for the taxon of 24% (range 17-31%) one year after the fire, assuming current management conditions (Legge et al. 2021). <u>High frequency fires</u> Frequent fire can decrease the availability of hollow-bearing trees in the landscape, and change the floristic composition and nutritional profile of glider habitat (Lindenmayer et al. 2013, Au et al. 2019). High frequency fire has reduced the density and stature of Mountain ash forests, posing threats to a range of tree-dependent fauna (Burns et al. 2015). In the Urbenville FMA of northern NSW, the species' abundance on survey sites was found to be significantly greater in forests that were infrequently burnt (Andrews et al. 1994). Too intense or frequent planned burning may contribute to population losses or declines in the southern part of the greater glider's range. Bluff (2016) reported that hollow-bearing trees (HBTs) affected by fire during planned burns were 28 times more likely to collapse than HBTs that were not burnt. Parnaby et al. (2010) found that following low intensity prescription burns in the Pilliga forests (NSW), mean collapse rates for burnt HBTs were 14-26%. This was consistent with the collapse rate of 25.6% found by Bluff (2016). A survey following a planned burn at Tallarook Range in the Central Highlands (Vic) in 2021 found that a large number of potential greater glider habitat trees were burnt, with "many destroyed" (N. Stimson 2021, pers. comm. 26 June). There is increased pressure from some parts of the community to undertake more hazard reduction burning, follow the severe bushfires of 2019-20. <u>Interactions with habitat clearing</u> Habitat fragmentation, due to clearing, can compound the impact of fires by hampering the ability of species to recolonise burnt areas (Dickman et al. 2020). Populations of greater gliders (southern and central)

Threat	Status and severity [a]	Evidence
		have disappeared after major bushfires; for example, no individuals were recorded for 19 years after a 1994 fire in the isolated Royal National Park (NSW) (Andrews et al. 2014). The impacts of fire on greater glider habitat are higher in landscapes that have been subject to previous timber harvesting, and at sites where post-fire salvage operations take place (Bowd EJ et al. 2021; Lindenmayer et al. 2021). Following the 2009 Victorian bushfires, 79% of large living trees with cavities died in the *Eucalyptus regnans* (Mountain Ash) forests, with no recruitment of new large cavity-bearing trees by 2011 (Lindenmayer et al. 2012). This was attributed to repeated past fires, and widespread timber harvesting which had resulted in the landscape being dominated by young stands. In the Dorrigo, Guy Fawkes and Chaelundi plateaux of north-eastern NSW, the combined effects of high fire frequency and high timber harvesting intensity resulted in greater declines of greater gliders (southern and central) than each threat alone (McLean et al. 2018). Physical disturbances associated with firefighting operations and post-fire 'mop up' include construction of roads and fire control lines, earthworks, tree removal and expansion of burnt areas through backburning (Driscoll et al. 2010). After fires, hazardous trees with large hollows are often felled for safety reasons (along roads, fire trails and walking trails) within greater glider habitat (DECCW 2011). Andrew (2001) reported that 120 HBTs were felled after the 1994 bushfires in Royal National Park by NPWS due to concerns about public safety. In Vic, loss of HBTs due to mechanical site preparation works associated with prescribed burning (which primarily occurs in foothill forests close to settled areas) may reduce suitable habitat for the greater glider (southern and central). Trees that are assessed as potentially hazardous (if they were to catch fire) are routinely removed from the perimeter of planned burns on public land in Vic. They are also removed from bushfire control lines during and after bushfire suppression activities (DELWP n.d). Although not all hazardous trees are hollow bearing, many are, or are likely to be trees that form hollows more quickly (J Nelson 2021. pers comm 16 April). <u>Interactions with climate change</u> Fire poses an increasing risk to the species. Indicators of forest fire danger in south-eastern Australia have emerged outside of the range of historical experience. More than 23% of the temperate forests in south-eastern Australia were burnt in the 2019-20 fire season, making the scale of the fires unprecedented both for Australia and globally (Boer et al. 2020). The radiative power of the 2019-20 fires, and the number of fires that developed into pyroconvective storms, were also unmatched in Australia's historical record (Boer et al. 2020). The multiple climate change contributors to fire risk in southeast Australia, as well as the observed non-linear escalation of fire extent and intensity, increase the likelihood that fire events will rapidly intensify in the future (Boer et al. 2020).

Threat	**Status and severity** [a]	**Evidence**
Habitat clearing and fragmentation	• Timing: current and future • Confidence: observed • Consequence: catastrophic • Trend: decreasing • Extent: across parts of the range	The greater glider is absent from cleared areas and has little dispersal ability to move through cleared areas between fragments (Tyndale-Biscoe & Smith 1969b; McCarthy & Lindenmayer 1999a,b; Lindenmayer et al. 2000; Eyre 2006; Taylor & Goldingay 2009). Population viability in small remnants is low due to the species' low reproductive output, susceptibility to disturbance and edge effects. Extensive land clearing for development and agriculture has led to fragmentation of habitat in some areas, e.g. the Tumut area of NSW (Pope et al. 2004) where small subpopulations exist in a pine matrix. About 30 years after clearing in the Tumut area, Lindenmayer et al. (1999) found that the occupancy rate of greater gliders (southern and central) in remnant patches was 21% compared to 38% in the surrounding native forest, indicating that recolonization does not occur readily. The probability of occurrence was significantly greater in large remnants, sites on flat terrain, and sites dominated by particular eucalypt forest types. Genetic analysis in the Tumut population by Taylor et al. (2007) indicated that some immigration into patches was occurring, with dispersal through distances of up to 7 km recorded, but there were lower levels of immigration and genetic mixing in patches further (> 1 km) from continuous forest. Artificial wildlife crossing structures to aid gliders to cross gaps such as highways and powerline easements have now been built within greater glider habitat throughout eastern Australia (Dalton 2018; Goldingay et al. 2018, 2020). Greater gliders have been recorded using these structures at only one location. At this site, the Sugarloaf Pipeline in Victoria, greater gliders were occasionally recorded on glide poles, although it is unclear whether they were using them to cross gaps or to move parallel to gaps (GHD 2017; Dalton 2018). The absence of records of greater gliders crossing highways or railways, despite glide poles being installed and monitored in multiple projects, suggest that they may be reluctant to cross near traffic. Gliders have rarely been recorded using nest boxes. In a 3-year study of 120 nest boxes in Victoria, one was used on two occasions by a greater glider, 640 days after placement (Menkhorst 1984). No greater glider use was seen in a 4-year study on 96 nest boxes (Lindenmayer et al. 2003). In a study of 206 glider-suitable nest boxes intended to mitigate loss of tree hollows from construction of 16 roads projects in NSW, 3 boxes in 2 projects were used by greater gliders (Goldingay et al. 2020). Nest boxes have been utilised in a 2-year program at Mirboo North (Gippsland, Vic) to increase habitat connectivity, with greater gliders (southern and central) filmed using the boxes as maternal dens (D Liepa 2020. pers comm 10 September). Alterations to improve the thermal properties of nest boxes may improve their use by greater gliders (e.g. Griffiths et al. 2018). Fragmentation effects are likely exacerbated by inappropriate fire regimes.

Timber harvesting	• Timing: current and future • Confidence: observed • Consequence: major • Trend: decreasing • Extent: across parts of the range	The sensitivity of greater gliders (southern and central) to timber harvesting has been well documented. Although some habitat across the species' range is found in conservation reserves (Smith & Smith 2018, Wagner et al. 2020), where timber harvesting is excluded and the removal of HBTs is subject to constraints, prime habitat coincides largely with areas suitable for timber harvesting (Braithwaite 1984). There is a progressive decline in numbers of HBTs in some production forests, as harvesting rotations become shorter and dead stags collapse, and HBTs are not being replaced due to lack of recruitment (Ross 1999; Ball et al. 1999; Lindenmayer et al. 2011, 2012). The degree of impact depends on forest type and timber harvesting intensity, with larger declines in more heavily logged sites (Tyndale-Biscoe & Smith 1969b; Lunney 1987; Kavanagh et al. 1995; Kavanagh & Webb 1998; Kavanagh 2000; McLean et al. 2018). In the Central Highlands of Vic, where clearfelling is undertaken, Lindenmayer et al. (2017b) found that the rate of loss of HBTs greatly exceeded the rate of recruitment. The area of clearfelled forest adjacent to wildlife corridors was also found to increase the chance of collapse of HBTs, possibly due to the greater exposure of stems to elevated wind speeds at corridor edges. However, models investigating the impacts of forest disturbance on the greater glider (southern and central) in the same area found that timber harvesting in the surrounding landscape was not a significant covariate influencing the probability of occurrence of the species (Lindenmayer et al. 2020). Recovery of subpopulations following timber harvesting is slow. Subpopulations in south-east NSW had not recovered 8 years after timber harvesting in sites retaining 62%, 52% and 21% of the original tree basal area (Kavanagh & Webb 1998). In the regrowth Mountain Ash forests (Central Highlands) of Vic, greater gliders (southern and central) were absent post-timber harvesting until the forests were >38 years old (Macfarlane 1988). Greater Gliders can persist, albeit likely in lower numbers, following harvesting. Kavanagh (2000) found that, in production forests in south-east NSW, subpopulations could persist post-timber harvesting if 40% of the original tree basal area was retained, provided (adjoining) riparian vegetation was also protected. An analysis overlaying all detections (from the Victorian Biodiversity Atlas and VicForests Species Observations layer) made post-harvest in timber harvesting areas in Vic since 1980, found that the species can persist in timber harvesting regrowth areas of very young age (VicForests 2021). The impacts of timber harvesting on greater gliders can be mitigated by landscape-level management strategies that retain habitat corridors and HBTs (Eyre 2006; Woinarski et al. 2014). In 2019, VicForests began moving away from clearfelling towards variable retention systems, which aim to retain more habitat trees and reduce the use of controlled burns for regeneration post-harvest. Protections for the species in East Gippsland and the Midlands (where Special Management Zones were required) were also revised to retain 40% of the basal area of eucalypts across each coupe where ≥5 greater gliders per km^2 are identified.

Threat	**Status and severity** [a]	**Evidence**
		Under the new Victorian Forestry Plan, harvest rates will reduce from 2024, leading up to a cessation of all native forest timber harvesting by 2030 (VicForests 2021). However, cumulative impacts of the 2019-20 bushfires, ongoing prescribed burning, timber harvesting and climate change will continue to put pressure on remaining greater glider habitat. Fire-logging interactions likely increase risks to greater glider populations.
Barbed wire fencing (entanglement)	• Timing: current and future • Confidence: observed • Consequence: minor • Trend: unknown • Extent: across the entire range	There are occasional losses of individuals due to entanglement in barbed wire fences across the greater glider's range (van der Ree 1999).
Climate Change		
Increased temperatures and changes to rainfall patterns	• Timing: current and future • Confidence: observed • Consequence: major • Trend: increasing • Extent: across the entire range	Mean temperatures across the distribution of greater glider have risen by 1.4 degrees and heat waves have become longer and more frequent over the past century (BOM & CSIRO 2020). In the southern part of the range, winter rainfall has declined by 12% since the 1990's, but summer rainfall remains unchanged. These trends are projected to continue over the coming decades under moderate and high emissions scenarios (CSIRO & BOM 2021). A unique physiology and a strict eucalypt diet make the greater glider vulnerable to high temperatures and low water availability (Rübsamen et al. 1984). Prolonged exposure to temperatures over 40°C is likely to lead to high mortality (Rübsamen et al. 1984). Moore et al. (2004) suggested that the preference of greater gliders for higher elevations is because they are sensitive to heat and must expend energy and considerable water to cool themselves when the ambient temperature is over 20°C. The increase of night-time temperatures has been implicated in the decline of greater glider (southern and central) numbers in Vic subpopulations (Wagner et al. 2020). At lower altitude (<500 m) surveyed sites in the Blue Mountains, increasing mean annual temperatures were attributed to be the cause of declines of greater gliders (southern and central), suggesting that night-time as well as day-time temperatures may be impacting the species, especially during heatwaves (Smith & Smith 2018, 2020). During extreme hot days over the 2019-2020 summer in the Blue Mountains and Lithgow LGA, two individuals were found on the ground and died soon after rescue. An autopsy concluded that they died as a result of drought and extreme heat (P Ridgeway 2021. pers comm 6 January). This further suggests that daytime temperatures are impacting the species. Water stress affects growth in forest eucalypts (Matusick at al. 2013) and reduces the availability of young, more palatable foliage. Combined with higher temperatures and extreme heat events this may cause heat stress, drought stress and mortality (Vic SAC

Threat	Status and severity [a]	Evidence
		2015). Elevated CO_2 may change the nutritional and water content of eucalypt leaves (Duan et al. 2019), though effects are difficult to predict and may have only a small impact on greater glider survival (Hovenden & Williams 2010). A warmer climate also reduces the nutritional and water content of eucalypt leaves (Foley et al. 1990; Lawler et al. 1997; Gleadow et al. 1998; McKiernan et al. 2014), and could be expected to reduce reproduction rates and population size (DeGabriel et al. 2009; Kearney et al. 2010). Above temperatures of 35°C, greater gliders need to dissipate >100% of metabolic heat production by evaporative means (Rübasamen 1984). At the same time, they reduce their food intake due to thermogenesis, leading to their energy and water stores being rapidly expended (Beale et al. 2018; Youngentob et al. 2021). This can lead to death of both young and adult gliders, or if less severe, can reduce growth in milk-fed young and reduce the health and fitness of adult gliders (Youngentob et al. 2021). Altered weather conditions are leading to higher frequency and intensity of bushfires (BOM & CSIRO 2020), further compounding the impacts of climate change on greater gliders. Large storms, particularly following fire or timber harvesting, may also result in the further loss of old hollow-bearing trees (Lindenmayer et al. 2018a). The age and dominant species of trees in the forests of east coast Australia are likely to continue to alter over the coming century, due to the compounding impacts of climate change, fire, clearing and timber harvesting. Some eucalypt species preferred by greater gliders may be lost from sites where they currently occur as conditions become climatically unsuitable for these trees (Butt et al. 2013; González-Orozco et al. 2016; Booth 2017). It difficult to robustly predict how and where forests will change, as local genetics, disturbance history, soil, topography, and hydrology can all influence how native forest respond to climate change (Booth et al. 2015; Booth 2018).
Over-abundant native species		
Hyper-predation by owls	• Timing: current and future • Confidence: observed • Consequence: moderate • Trend: static • Extent: across parts of the range	The greater glider forms a significant part of the diet of *Ninox strenua* (powerful owl) (Bilney et al. 2006), and has become a significant part of the diet of *Tyto tenebricosa* (sooty owl) since European settlement due to the widespread decline of terrestrial prey species for these owls (Bilney et al. 2010). The greater glider has significantly declined or become locally extinct in some intact forest areas, possibly due to owl predation (Lindenmayer et al. 2011, 2018b; P. Rickards pers. comm. 2015). At one site over a three-year period, two powerful owls were suspected to have reduced a greater glider (southern and central) population from 80 to 7 individuals (Kavanagh 1988). Hyper-predation by large forest owls may possibly be due to increased abundance of owls following release from competition with the European red fox for prey, caused in turn by suppression of red fox populations by baiting activities (Lindenmayer et al. 2011). However, the presence of large forest owls does not necessarily indicate a population-level impact on

Threat	Status and severity [a]	Evidence
		greater gliders. Numbers of powerful and sooty owls have increased greatly in the Blue Mountains since the 1980s and these species have been recorded at many sites with greater gliders, but no significant relationship between greater glider abundance and the presence of either owl species was found (Smith & Smith 2018). Effects may be exacerbated by fire-predator interactions.
Competition from *Cacatua galerita* (Sulphur-crested Cockatoos)	• Timing: current and future • Confidence: suspected • Consequence: minor • Trend: increasing • Extent: across parts of the range	Numbers of Sulphur-crested Cockatoos in the Blue Mountains have increased significantly since 1990 and may be competing with greater gliders for hollows. They have been observed taking over nesting hollows of powerful owls and have been roosting in increasing numbers at several greater glider sites since 2007 (Smith & Smith 2018). However, no significant relationship was found between greater glider (southern and central) abundance and the number of roosting cockatoos (Smith & Smith 2018). Further research is required to determine the impact of inter-species competition for hollows on greater gliders.
Introduced species		
Predation by feral cats (*Felis catus*)	• Timing: current and future • Confidence: observed • Consequence: minor • Trend: unknown • Extent: across the entire range	Remains of greater gliders have been found in the stomachs of feral cats, however they formed a tiny proportion of the overall animals consumed (Jones & Coman 1981). It is unclear whether they were killed by cats (if so, most likely when gliders come to the ground) or consumed as carrion. After wildfires, greater gliders are displaced and have been observed on the ground where they are more susceptible to predation (Fleay 1947), suggesting that fire-predator interactions amplify threats to the species.
Predation by European red foxes (*Vulpes vulpes*)	• Timing: current and future • Confidence: observed • Consequence: minor • Trend: unknown • Extent: across the entire range	Remains of greater gliders have been found in the stomachs and scats of European red foxes (Coman 1973; Brunner et al. 1975; Wallis & Brunner 1986; Lunney et al. 1990). However, they formed a tiny proportion of the overall animals consumed, and it is unclear whether they were killed by foxes (if so, most likely when gliders come to the ground) or consumed as carrion. After wildfires, greater gliders are displaced and have been observed on the ground where they are more susceptible to predation (Fleay 1947), suggesting that fire-predator interactions amplify threats to the species.

Timing—identify the temporal nature of the threat;

Confidence—identify the extent to which we have confidence about the impact of the threat on the species;

Consequence—identify the severity of the threat;

Trend—identify the extent to which it will continue to operate on the species;

Extent—identify its spatial content in terms of the range of the species.

Each threat has been described in Table 1 in terms of the extent that it is operating on the species. 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 Greater glider (southern and central) risk matrix

Likelihood	Consequences				
	Not significant	**Minor**	**Moderate**	**Major**	**Catastrophic**
Almost certain	Low risk	Moderate risk	Very high risk	Very high risk **Timber harvesting** **Increased temperatures and changes to rainfall patterns**	Very high risk **Inappropriate fire regimes** **Habitat clearing and fragmentation**
Likely	Low risk	Moderate risk **Competition from Sulphur-crested Cockatoos**	High risk	Very high	Very high risk
Possible	Low risk	Moderate risk	High risk **Hyper-predation by owls**	Very high risk	Very high risk
Unlikely	Low risk	Low risk **Predation by foxes** **Predation by feral cats** **Barbed wire fencing (entanglement)**	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 objective

Within the next three generations, the population size as well as the extent, quality and connectivity of habitat required to maintain the population will have increased.

Conservation and management priorities

Habitat loss, disturbance and modification (including fire)

- In the aftermath of bushfires, protect any unburnt habitat (within or adjacent to recently burnt landscapes) in order to support population recovery. This includes, but is not limited to:
    - Areas identified to be important post-fire refuges.
    - Protecting hollow-bearing trees from post-fire salvage timber harvesting and clean-up operations.
    - Avoiding hazard reduction burns in these areas.
- Re-assess and revise current prescriptions used for prescribed burning to ensure that the frequency and severity of fires in greater glider habitat are minimised, in order to mitigate the risk of further population declines and loss of hollow-bearing trees. Measures to reduce risk from future bushfires should be strategic, incorporate adaptive management, and include a risk assessment that considers trade-offs between fire control efficiency and environmental damage.
- Implement and enforce measures to reduce direct mortality and loss of hollow-bearing trees during site preparation and execution of prescribed burns, including rake hoeing around the base of trees.
- Ensure that eucalypt forests and the impacts of disturbance (including fire) are managed to prevent them transitioning to less nutritious, hotter, and/or more fire-prone plant communities, and to ensure that food tree species preferred by the greater glider (southern and central) continue to be the dominant canopy trees.
- Protect and maintain sufficient areas of suitable habitat, including denning and foraging resources and habitat connectivity, to sustain viable subpopulations throughout the species' range.
- Protect hollow-bearing trees on private property, roadside reserves, and along the edges of roads/tracks. Prior to removing trees identified to be a 'hazard', undertake a risk assessment by a suitably qualified person to determine whether their removal is necessary, including a consideration of the potential impacts of tree removal on the greater glider. Incorporate measures to ensure ongoing recruitment of hollow-bearing trees into planning processes.
- Avoid fragmentation and loss of habitat due to development of new transport corridors. Include consideration of the species in planning processes, and where possible re-locate recreational activities and roads away from habitat.
- Establish, maintain and enforce effective prescriptions in production forests to support populations of the greater glider (southern and central). This includes, but is not limited to: appropriate levels of habitat retention, timber harvesting exclusion and timber harvesting

rotation cycles; maintenance of wildlife corridors between harvested patches; maintenance of vegetation buffers around habitat patches excluded from harvesting; protection of existing hollow-bearing trees with appropriate buffers; adequate recruitment of hollow-bearing trees; maintaining preferred food tree species as dominant canopy trees; and minimal use and adequate containment of regeneration burns. Clearfelling should be avoided, as well as timber harvesting in climate or post-fire refuges.

- As a last resort, where hollows are limiting, consider the use of nest boxes and artificial hollows that are suitable for the species. Monitor use of these structures to ensure they are being utilised, and revise designs or placement as required.
- Restore habitat and connectivity:
    - where habitat has been substantially fragmented, disturbed or modified,
    - between small habitat patches and larger areas of contiguous forest,
    - at a landscape scale through projects such as the Great Eastern Ranges Initiative, to facilitate movement and recolonisation of areas impacted by fires, droughts or other factors, and to provide opportunities for the species to adapt to the changing climate,
    - following climate-ready restoration guidelines (e.g. Hancock et al. 2018), and
    - following the National Restoration Standards (Standards Reference Group SERA 2021).
- Revise mitigation and offset guidelines for development and linear infrastructure (e.g. pipelines, transport corridors) to reflect the limited effectiveness of artificial structures (nest boxes, glide poles) as mitigation actions for loss, degradation or fragmentation of greater glider habitat.
- Avoid the use of barbed wire, and replace the top strands of existing barbed wire with single-strand wire in habitat known to be occupied by greater gliders.

Climate change

- Protect all habitat likely to be climate change refuges, including sites buffered against desiccating conditions (e.g. sheltered and/or on south-facing aspects), under future climate change scenarios. Where possible, maintain or establish connectivity with other habitat in order to facilitate movement.
- Undertake habitat restoration to improve micro-climate conditions in areas at high risk of extreme temperatures and drought.

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

- Where threats from introduced predators (including the European red fox and feral cat) are locally significant:
    - Implement appropriate control measures, particularly in areas burnt by bushfires.
    - Develop and implement longer-term strategies to control predation by the European red fox and feral cat, as detailed in the relevant Threat Abatement Plans.

Ex-situ recovery actions

- Investigate the feasibility of reintroductions to areas from which the species has recently been extirpated, where natural recolonisation is unlikely.

- If feasible, undertake translocations to these areas, ensuring that habitats are managed for future suitability including adaptive management of threats that may have led to the species' extirpation.
- Ensure that any proposals for translocations are developed collaboratively and focused on the best conservation outcomes for the species.

Stakeholder engagement/community engagement

- Seek stakeholder input into assessment and planning processes that include protections for the greater glider (southern and central) and its habitat. This may include environmental impact assessments, park management plans, water resource plans, fire management plans and transport development plans.
- Develop and implement a communication strategy around the need to balance hazard reduction burning with the need to conserve and protect species and habitats.
- Liaise with private landholders, Traditional Owners, and conservation and land management groups to co-create guidelines for on-ground management of the greater glider (southern and central).
- Support volunteer involvement in surveying and monitoring, in particular gathering data on the species' occurrence and foraging habitat, and in the implementation of conservation actions.
- Pursue opportunities with landholders to enter land management agreements, particularly in-perpetuity covenants, that promote the protection and maintenance of habitat on private lands with high value for the species.
- Engage and involve Traditional Owners in conservation actions, including survey, monitoring and management actions.
- Foster public interest in the species and its ongoing conservation, to increase support for the implementation of conservation actions.

Survey and monitoring priorities

- Implement an integrated long-term monitoring program across the species' range to:
    - determine trends in abundance and distribution,
    - ascertain the status and viability of subpopulations,
    - assess the impacts of compounding threats, and
    - evaluate the relative benefits and effectiveness of management actions.
- Conduct on-ground surveys to establish habitat and population impacts as a result of the 2019–20 bushfires and to provide a baseline for future population monitoring. Leverage post-fire monitoring at sites where surveys were undertaken prior to 2019–20, to assess population trends across the fire cycle. Undertake these actions for any future large-scale events such as bushfires, heatwaves or drought.
- Monitor the incidence and impacts of fire and timber harvesting in the species' range, particularly in areas adjacent to those burnt in the 2019–20 bushfires.

- Monitor the abundance, age and size structure of hollow-bearing trees and their responses to management measures. This includes before and after prescribed burns, and before and after timber harvesting.
- Continue to undertake surveys on high priority timber harvesting coupes as part of DELWP's Forest Protection Survey Program (begun in 2018), and other pre-harvest surveys, to inform adaptive management in timber harvesting areas.

Information and research priorities

- Undertake genetic sampling to resolve taxonomy, especially in areas where there is contact between the two greater glider species and subspecies.
- Improve understanding of actions that can be undertaken to improve rates of survival and recovery following major bushfires (including characteristics of refuges, role of patchiness in fire severity, and interactions with habitat quality and disturbance history).
- Support the development of guidelines for fire management by assessing the impacts of fire management and different fire regimes (including frequency and intensity) on habitat, subpopulation size and hollow availability.
- Define appropriate levels of timber harvesting exclusion, and hollow-bearing tree retention and recruitment, to maintain subpopulation sizes and persistence across the species' distribution. Assess and monitor the species' response to current timber harvesting prescriptions and revise as required, noting that the effectiveness of prescriptions may differ on a regional basis depending on forest type.
- To support protection and restoration activities, improve understanding of the species' behaviours, and landscape and habitat features, that promote or constrain genetic and functional connectivity between greater glider habitat patches.
- Investigate ways to improve the effectiveness of artificial structures for mitigation of impacts on greater gliders. Research should aim to evaluate effectiveness at a scale likely to be significant for subpopulation-level recovery rather than isolated instances of use (e.g. genetic connectivity provided by glide poles over transport routes, feasibility of artificial hollows and nest boxes to sustain populations).
- Investigate the impact of inter-species competition for hollows on the greater glider, and the extent to which this may be inhibiting subpopulation recovery.
- Investigate changes in subpopulations or dietary preferences of large owls, factors which may contribute to these changes, and the extent to which they may affect greater glider subpopulations.
- Improve understanding of actions that can be undertaken to improve rates of survival and recovery in climate-affected subpopulations.
- Identify areas likely to be climate refuges for the species under robust scenarios of climate change.
- Improve understanding of the species' diet and life history, especially in areas where subpopulations have declined. Determine the likely effects of increased temperatures and drought on food supply, behaviour and survival.

Recovery plan

The Committee recommends that there should be a recovery plan for *Petauroides volans* (greater glider (southern and central)). Stopping decline and supporting recovery is complex, due to a need to fully identify all the threats, the requirement for a high level of planning to abate the threats, a high level of support by key stakeholders, a high level of prioritisation and a highly adaptive management process. Existing mechanisms are not adequate to address these needs.

Links to relevant implementation documents

NSW Saving Our Species Strategy: Greater Glider Population in the Eurobadalla local government area (*Petauroides volans*) – Endangered Population)

NSW Saving Our Species Strategy: Greater Glider Population at Seven Mile Beach National Park (*Petauroides volans* – Endangered Population)

NSW Saving Our Species Strategy: Greater Glider Population at Mount Gibraltar Reserve area (*Petauroides volans* – Endangered Population)

Threat abatement plan for predation by feral cats 2015

Threat abatement plan for predation by the European red fox 2008

Threat abatement plan for predation by the European red fox 2008 - background document

Conservation Advice and Listing Assessment references

References cited in the advice

Andrew D (2001) *Post Fire Vertebrate Survey – Royal and Heathcote National Parks and Garrawarra State Recreation Area*. A report to NSW National Parks and Wildlife Service, Sydney south region.

Andrew D, Koffel D, Harvey G, Griffiths K & Fleming M (2014) Rediscovery of the greater glider *Petauroides volans* (Marsupialia: Petauroidea) in the Royal National Park, NSW. *Australian Zoologist* 37, 23–28.

Andrews SP, Gration G, Quin D & Smith AP (1994) Description and assessment of forestry impacts on fauna of the Urbenville Forestry Management Area. Report for State Forests of New South Wales. Austeco Environmental Consultants, Armidale.

Au J, Clark RG, Allen C, Marsh KJ, Foley WJ & Youngentob KN (2019) A nutritional mechanism underpinning folivore occurrence in disturbed forests. *Forest Ecology and Management* 453. Available on the internet at: https://doi.org/10.1016/j.foreco.2019.117585

Ball IR, Possingham HP & Lindenmayer DB (1999) A tree hollodynamics simulation model. *Forest Ecology and Management* 123, 179–194.

Beale PK, Marsh KJ, Foley WJ & Moore BD (2018) A hot lunch for herbivores: physiological effects of elevated temperatures on mammalian feeding ecology. *Biol Rev Camb Philos Soc* 93, 674–692.

Berry LE, Driscoll DA, Banks SC & Lindenmayer DB (2015) The use of topographic fire refuges by the greater glider (*Petauroides volans*) and the Mountain Brushtail Possum (*Trichosurus cunninghami*) following a landscape-scale fire. *Australian Mammalogy* 37, 39–45.

Bilney R, Cooke R & White J (2006) Change in the diet of sooty owls (*Tyto tenebricosa*) since European settlement: from terrestrial to arboreal prey and increased overlap with powerful owls. *Wildlife Research* 33, 17–24.

Bilney R, Cooke R & White J (2010) Underestimated and severe: Small mammal decline from the forests of south-eastern Australia since European settlement, as revealed by a top-order predator. *Biological Conservation* 143, 52–59.

Bluff L (2016) *Reducing the effect of planned burns on hollow-bearing trees*. Fire and adaptive management report no. 95. Victorian Government Department of Environment, Land, Water and Planning. Melbourne.

Boer MM, de Dios VR & Bradstock RA (2020) Unprecedented burn area of Australian mega forest fires. *Nature Climate Change* 10, 171–172.

Booth TH, Broadhurst LM, Pinkard E, Prober SM, Dillon SK, Bush D, Pinyopusarerk K, Doran JC, Ivkovich M & Young AG (2015) Native forests and climate change: Lessons from eucalypts. *Forest Ecology and Management* 347, 18–29.

Booth TH (2017) Impacts of climate change on eucalypt distributions in Australia: an examination of a recent study. *Australian Forestry* 80, 208–15.

Booth TH (2018) Species distribution modelling tools and databases to assist managing forests under climate change. *Forest Ecology and Management* 430, 196–203.

Bowd EJ, McBurney L, Blair DP & Lindenmayer DB (2021) Temporal patterns of forest seedling emergence across different disturbance histories. *Ecology and Evolution* 11, 9254–9292.

Braithwaite lW (1983) Studies on the arboreal marsupial fauna of eucalypt forest being harvested for woodpulp at Eden, New South Wales. I. The species and distribution of animals. *Australian Wildlife Research* 10, 219–229.

Braithwaite LW (1984) On identifying important habitat characteristics and planning a conservation strategy for arboreal marsupials within the Eden Woodpulp Concession area In *Possums and Gliders* (eds AP Smith & ID Hume), pp. 501–508. Surrey Beatty and Sons, Chipping Norton.

Brunner H, Loyd JW & Coman BJ (1975) Fox scat analysis on a forest park in south-eastern Australia. *Australian Wildlife Research* 2, 147–154.

Burns PA & Atkins ZS (2021) *Gliding from the ashes: Post-fire surveys for the greater glider and yellow-bellied glider in Far East Gippsland.* Snowline Ecology & Native Mouse Ecological Consulting.

Butt N, Pollock LJ & McAlpine CA (2013) Eucalypts face increasing climate stress. *Ecology and Evolution* 3, 5011–5022.

Chia EK, Bassett M, Nimmo DG, Leonard SWJ , Ritchie E, Clarke MF & Bennett AF (2015) Fire severity and fire-induced landscape heterogeneity affect arboreal mammals in fire-prone forests. *Ecosphere* 6. Available on the internet at: https://doi.org/10.1890/ES15-00327.1

Coman BJ (1973) The diet of red foxes, *Vulpes vulpes* L., in Victoria. *Australian Journal of Zoology* 21, 391–401.

BOM (Bureau of Metereology) & CSIRO (2020). *State of the Climate 2020.* Sixth edition. Available on the internet at: State of the Climate 2020: Bureau of Meteorology (bom.gov.au)

CSIRO & BOM (Bureau of Metereology) (2021) Climate Change in Australia: Regional Climate Change Explorer. Retrieved 17 August 2021 from: https://www.climatechangeinaustralia.gov.au/en/projections-tools/regional-climate-change-explorer/super-clusters/.

Dalton K (2018) *Use and effectiveness of glider pole linkages as a mitigation measure: Sugarloaf Pipeline project.* Australasian Network for Ecology and Transportation Conference, Creswick, Victoria.

Davey SM (1984) Habitat preferences of arboreal marsupials within a coastal forest in southern New South Wales, in AP Smith & ID Hume (eds), *Possums and Gliders.* Australian Mammal Society, Sydney. pp. 509–516.

DAWE (2021) Area of Occupancy and Extent of Occurrence for *Petauroides volans*. Unpublished report. Department of the Environment (Commonwealth), Canberra.

DECCW (Department of Environment Climate Change and Water) (2011) *The Vertebrate Fauna of Royal & Heathcote National Parks and Garrawarra State Conservation Area.* Department of Environment Climate Change and Water (NSW), Hurstville.

DeGabriel JL, Moore BD, Foley WJ & Johnson CN (2009) The effects of plant defensive chemistry on nutrient availability predict reproductive success in a mammal. *Ecology* 90, 711–719.

DELWP (Department of Environment, Land, Water and Planning) (n.d) *Forest Fire Management Victoria, Hazardous tree removal after bushfire.* Available on the internet at: https://www.ffm.vic.gov.au/__data/assets/pdf_file/0015/21282/Hazardous-tree-removal-after_KP.pdf

DELWP (Department of Environment, Land, Water and Planning) (2016) *Preliminary recommendation on a nomination for listing:* Petauroides volans volans. Flora and Fauna Guarantee – Scientific Advisory Committee. Department of Environment, Land, Water and Planning (Vic), Melbourne.

Dickman C, Driscoll D, Garnett S, Keith D, Legge S, Lindenmayer D, Maron M, Reside A, Ritchie E, Watson J, Wintle B & Woinarski J (2020) *After the catastrophe: a blueprint for a conservation response to large-scale ecological disaster*. Threatened Species Recovery Hub, January 2020. Available on the internet at: https://www.nespthreatenedspecies.edu.au/publications-and-tools/after-the-catastrophe-a-blueprint-for-a-conservation-response-to-large-scale-ecological-disaster

DoEE (2016a) *Conservation advice for* Petauroides volans *(greater glider).* Department of the Environment and Energy (Commonwealth), Canberra.

DoEE (2016b) *Greater glider* (Petauroides volans) *recovery plan workshop: Summary of outcomes*. Unpublished report. Department of the Environment and Energy (Commonwealth), Canberra.

Downes SJ Handasyde KA & Elgar MA (1997) The use of corridors by mammals in fragmented Australian eucalypt forests. *Conservation Biology* 11, 718–726.

DPI (Department of Primary Industries) (2020) *Drought in NSW*. Viewed: 14 September 2020. Available on the internet at: https://www.dpi.nsw.gov.au/emergencies/droughthub_old/drought-in-nsw

DPIE (Department of Planning, Industry and Environment) (2020) *Understanding the effects of the 2019-20 fires.* Viewed: 15 September 2020. Available on the internet at: https://www.environment.nsw.gov.au/topics/fire/park-recovery-and-rehabilitation/recovering-from-2019-20-fires/understanding-the-impact-of-the-2019-20-fires

Driscoll DA, Lindenmayer DB, Bennett AF, Bode M, Bradstock RA, Cary GJ, Clarke MF, Dexter N, Fensham R, Friend G, Gill M, James S, Kay G, Keith DA, MacGregor C, Possingham HP, Russell-Smith J, Salt D, Watson JEM, Williams D & York A (2010) Resolving conflicts in fire management using decision theory: asset-protection versus biodiversity conservation. *Conservation Letters* 3, 215–223.

Duan H, Ontedhu J, Milham P, Lewis JD & Tissue DT (2019) Effects of elevated carbon dioxide and elevated temperature on morphological, physiological and anatomical responses of *Eucalyptus tereticornis* along a soil phosphorus gradient. *Tree Physiology* 39, 1821–1837.

Eyre TJ (2002) Habitat preferences and management of large gliding possums in southern Queensland. Ph.D. thesis, Southern Cross University, Lismore.

Eyre TJ (2004) Distribution and conservation status of the possums and gliders of southern Queensland, in RL Goldingay and SM Jackson (eds) *The Biology of Australian Possums and Gliders*. Surrey Beatty and Sons Chipping Norton, New South Wales. pp. 1–25.

Eyre TJ (2005) Hollow-bearing trees in a coastal forest in south-east Queensland, Australia: Abundance, spatial distribution and management. *Pacific Conservation Biology* 11, 23–37.

Eyre TJ (2006) Regional habitat selection by large gliding possums at forest stand and landscape scales in southern Queensland, Australia. I. Greater glider (*Petauroides volans*). *Forest Ecology and Management* 235, 270–282.

Eyre TJ, Butler DW, Kelly AL & Wang J (2010) Effects of forest management on structural features important for biodiversity in mixed-age hardwood forests in Australia's subtropics. *Forest Ecology and Management* 259, 534–546.

Eyre TJ, Smith GC, Venz MF, Mathieson MT, Hogan LD & Starr C (2021) *Species specific advice: Greater gliders, Queensland*. Queensland Herbarium, Department of Environment and Science, Queensland Government. Brisbane.

Ferguson DJ, Laidlaw MJ & Eyre TJ (2018) Greater Glider Habitat Resource Assessment in the Burnett Mary. Department of Environment and Science (Qld).

Fleay D (1947) *Gliders of the Gum Trees*. Bread and Cheese Club, Melbourne.

Foley WJ, Kehl JC, Nagy KA, Kaplan IR. & Boorsboom AC (1990) Energy and water metabolism in free-living greater gliders *Petauroides volans. Australian Journal of Zoology* 38, 1–10.

GHD (2017) *Report for Melbourne Water Corporation - Sugarloaf Pipeline Project Toolangi Habitat Linkage Monitoring.* GHD. 31/29843.

Gibbons P & Lindenmayer DB (2002) *Tree hollows and wildlife conservation in Australia.* CSIRO Publishing, Collingwood.

Gleadow RM, Foley WJ & Woodrow IE (1998) Enhanced CO_2 alters the relationship between photosynthesis and defense in cyanogenic *Eucalyptus cladocalyx*. *Plant, Cell and Environment* 21, 12–22.

Goldingay RL (2012) Characteristics of tree hollows used by Australian arboreal and scansorial mammals. *Australian Journal of Zoology* 59, 277–294.

Goldingay RL, Taylor BD & Parkyn JL (2018) Use of tall wooden poles by four species of gliding mammal provides further proof of concept for habitat restoration. *Australian Mammalogy* 41, 255–261.

Goldingay RL, Rohweder D & Taylor BD (2020) Nest box contentions: Are nest boxes used by the species they target? *Ecological Management & Restoration* 21, 115–122.

González-Orozco CE, Pollock, Laura J, Thornhill, Andrew H, Mishler, Brent D, Knerr, N, Laffan, Shawn W, Miller, Joseph T, Rosauer, Dan F, Faith, Daniel P, Nipperess, David A, Kujala, H, Linke, S, Butt, N, Külheim, C, Crisp, Michael D & Gruber B (2016) Phylogenetic approaches reveal biodiversity threats under climate change. *Nature Climate Change* 6, 1110–1114.

Griffiths SR, Lentini PE, Semmens K, Watson SJ, Lumsden LF & Robert KA (2018) Chainsaw-Carved Cavities Better Mimic the Thermal Properties of Natural Tree Hollows than Nest Boxes and Log Hollows *Forests* 9, 235. Available on the internet at: https://doi.org/10.3390/f9050235

Hancock N, Harris R, Broadhurst L & Hughes L (2018) Climate-ready revegetation. A guide for natural resource managers. Version 2. Sydney, Macquarie University. Available on the internet at: https://www.mq.edu.au/__data/assets/pdf_file/0006/807666/Climate-Reveg-Guide-v2-2018.pdf

Henry SR (1984) Social organisation of the greater glider (*Petauroides volans*) in Victoria, in AP Smith & ID Hume (eds), *Possums and Gliders.* Surrey Beatty and Sons, Chipping Norton. pp. 221–228.

Henry SR (1995) Greater glider *Petauroides volans*, in PW Menkhorst (ed), *Mammals of Victoria: distribution, ecology and conservation.* Oxford University Press, Melbourne. pp. 118–120.

Hovenden MJ & Williams AL (2010) The impacts of rising CO2 concentrations on Australian terrestrial species and ecosystems. *Austral Ecology* 35, 665–684.

Incoll RD, Loyn RH, Ward SJ, Cunningham RB & Donnelly CF (2001) The occurrence of gliding possums in old-growth forest patches of mountain ash (*Eucalyptus regnans*) in the Central Highlands of Victoria. *Biological Conservation* 98, 77–88.

IUCN Standards and Petitions Committee (2019) *Guidelines for Using the IUCN Red List Categories and Criteria. Version 14.* Prepared by the Standards and Petitions Committee. Available on the internet at: http://www.iucnredlist.org/documents/RedListGuidelines.pdf

Jackson S & Groves C (2015) *Taxonomy of Australian mammals*. CSIRO Publishing, Clayton South.

Jones E & Coman BJ (1981) Ecology of the Feral Cat, Felis *catus* (L.), in South-Eastern Australia I. Diet. *Wildlife Research* 8, 537–547.

Kavanagh RP (1984) Seasonal changes in habitat use by gliders and possums in southeastern New South Wales. In AP Smith ID Hume (eds) *Possums and Gliders*. Surrey Beatty and Sons, Chipping Norton. pp. 527–543.

Kavanagh RP (1987) *Floristic and phenological characteristics of a eucalypt forest in relation to its use by arboreal marsupials*. MSc thesis. Australian National University, Canberra.

Kavanagh RP (1988) The impact of predation by the powerful owl, *Ninox strenua*, on a population of the greater glider, *Petauroides volans*. *Austral Ecology* 13, 445–450.

Kavanagh RP (1995) *Nocturnal Forest Birds and Arboreal Marsupials of Coolah Tops, Warung Management Area, Western Region*. Forest Resources Series No. 28. State Forests of New South Wales, Sydney. 24 pp.

Kavanagh RP (2000) Effects of variable-intensity logging and the influence of habitat variables on the distribution of the greater glider *Petauroides volans* in montane forest, southeastern New South Wales. *Pacific Conservation Biology* 6, 18–30.

Kavanagh RP (2004) Distribution and conservation status of possums and gliders in New South Wales. In RL Goldingay & SM Jackson (eds) *The Biology of Australian Possums and Gliders*. Surrey Beatty and Sons, Chipping Norton. pp. 130–148.

Kavanagh RP & Bamkin KL (1995) Distribution of nocturnal forest birds and mammals in relation to the logging mosaic in south-eastern New South Wales, Australia. *Biological Conservation* 71, 41–53.

Kavanagh RP, Debus S, Tweedie T & Webster R (1995) Distribution of nocturnal forest birds and mammals in north-eastern New South Wales: relationships with environmental variables and management history. *Wildlife Research* 22, 359–377.

Kavanagh RP & Lambert M (1990) Food selection by the greater glider: is foliar nitrogen a determinant of habitat quality? *Australian Wildlife Research* 17, 285–299.

Kavanagh RP & Webb GA (1998) Effects of variable-intensity logging on mammals, reptiles and amphibians at Waratah Creek, south-eastern New South Wales. *Pacific Conservation Biology* 4, 326–347.

Kavanagh RP & Wheeler RJ (2004) Home range of the greater glider *Petauroides volans* in tall montane forest of southeastern New South Wales, and changes following logging. In RL Goldingay & SM Jackson (eds) *The Biology of Australian Possums and Gliders*. Surrey Beatty and Sons, Sydney. pp. 413–425.

Kavanagh R, McLean C & Stanton M (2021) *Greater glider and yellow-bellied glider population responses to the 2019-2020 wildfires in north-eastern NSW and south-eastern NS*W. A report to the Department of Agriculture, Water and Environment. Canberra.

Kearney MR, Wintle BA & Porter WP (2010) Correlative and mechanistic models of species distribution provide congruent forecasts under climate change. *Conservation Letters* 3, 203–213.

Kehl J & Borsboom A (1984) Home range, den tree use and activity patterns in the greater glider (*Petauroides volans*). In AP Smith ID Hume (eds) *Possums and Gliders*. Surrey Beatty and Sons, Chipping Norton. pp. 229–236.

Kerr R (1792) *The animal kingdom, or zoological system, of the celebrated Sir Charles Linnaeus: Class 1. Mammalia: containing a complete sydtematic description, arrangement, and nomenclature, of all known species and varieties of the Mammalia, or animals which give suck to their young: being a translation of that part of the Systema Naturae, as lately published, with great improvements, by Proffessor Gmelin of Goettingrn. Together with numerous additions from more recent zoological writers, and illustrated with copperplates.* J Murray & R Faulder: London.

Land Conservation Council (1984) *North-eastern area (Benalla-Upper Murray) Review*. Melbourne.

Lawler IR, Foley WJ, Woodrow IE & Cork SJ (1997) The effects of elevated CO_2 atmospheres on the nutritional quality of *Eucalyptus* foliage and its interaction with soil nutrient and light availability. *Oecologia* 109, 59–68.

Legge S, Woinarski JCZ, Garnett ST, Nimmo D, Scheele BC, Lintermans M, Whiterod N & Ferris J (2020) *Rapid analysis of impacts of the 2019–2020 fires on animal species, and prioritisation of species for management response*. Report prepared for the Wildlife and Threatened Species Bushfire Recovery Expert Panel. 14 March 2020. Department of Agriculture, Water and the Environment (Commonwealth), Canberra.

Legge S, Woinarski JCZ, Garnett ST, Geyle H, Lintermans M, Nimmo DG, Rumpff L, Scheele BC, Southwell DG, Ward M, Whiterod NS, Ahyong ST, Blackmore CJ, Bower DS, Brizuela-

Torres D, Burbidge AH, Burns PA, Butler G, Catullo R, Dickman CR, Doyle K, Ensbey M, Ehmke G, Ferris J, Fisher D, Gallagher R, Gillespie GR, Greenlees MJ, Hayward-Brown B, Hohnen R, Hoskin CJ, Hunter D, Jolly C, Kennard M, King A, Kuchinke D, Law B, Lawler I, Lawler S, Loyn R, Lunney D, Lyon J, MacHunter J, Mahony M, Mahony S, McCormack RB, Melville J, Menkhorst P, Michael D, Mitchell N, Mulder E, Newell D, Pearce L, Raadik TA, Rowley J, Sitters H, Spencer R, Valavi R, Ward M, West M, Wilkinson DP & Zukowski S (2021) *Estimates of the impacts of the 2019-2020 fires on populations of native animal species*. NESP Threatened Species Recovery Hub project 8.3.2 report. Brisbane, Australia.

Lindenmayer DB (2002) *Gliders of Australia: A Natural History.* UNSW Press, Kensington.

Lindenmayer DB (2009) *Forest pattern and ecological process: a synthesis of 25 years of research*. CSIRO Publishing, Melbourne.

Lindenmayer DB & Sato C (2018) Hidden collapse is driven by fire and logging in a socioecological forest ecosystem. *Procedures of the National Academy of Sciences* 115, 5181–5186.

Lindenmayer DB, Cunningham RB, Tanton MT, Smith AP & Nix HA (1990a). Habitat requirements of the mountain brushtail possum and the greater glider in the montane ash-type eucalypt forests of the central highlands of Victoria. *Australian Wildlife Research* 17, 467–478.

Lindenmayer DB, Cunningham RB, Tanton MT & Smith AP (1990b) The conservation of arboreal marsupials in the montane ash forests of the Central Highlands of Victoria, southeast Australia. I. Factors affecting the occupancy of trees with hollows. *Biological Conservation* 54, 111–131.

Lindenmayer DB, Cunningham RB, Tanton MT, Smith AP & Nix HA (1991) Characteristics of hollow-bearing trees occupied by arboreal marsupials in the montane ash forests of the Central Highlands of Victoria, south-east Australia. *Forest Ecology and Management* 40, 289–308.

Lindenmayer DB, Cunningham RB & Donnelly CF (1993) The conservation of arboreal marsupials in the montane ash forests of the Central Highlands of Victoria, southeast Australia. IV. The distribution and abundance of arboreal marsupials in retained linear strips (wildlife corridors) in timber production forests. *Biological Conservation* 66, 207-221.

Lindenmayer DB, Cunningham RB, Pope M & Donnelly CF (1999) The response of arboreal marsupials to landscape context: a largescale fragmentation study. *Ecological Applications* 9, 594–611.

Lindenmayer DB, Lacy RC & Pope ML (2000) Testing a simulation model for population viability analysis. *Ecological Applications* 10, 580–597.

Lindenmayer DB, Ball I, Possingham HP, McCarthy M & Pope ML (2001) A landscape test of the predictive ability of a spatially explicit model for population viability analysis. *Journal of Applied Ecology* 38, 36–48.

Lindenmayer DB, MacGregor CI, Cunningham RB, Incoll RD, Crane M, Rawlins D & Michael DR (2003) The use of nest boxes by arboreal marsupials in the forests of the central highlands of Victoria *Wildlife Research* 30, 259–264.

Lindenmayer DB, Pope ML & Cunningham RB (2004). Patch use by the greater glider (*Petauroides volans*) in a fragmented forest ecosystem. II. Characteristics of den trees and preliminary data on den-use patterns. *Wildlife Research* 31, 569–577.

Lindenmayer DB, Wood JT, McBurney L, MacGregor C, Youngentob K & Banks SC (2011) How to make a common species rare: a case against conservation complacency. *Biological Conservation* 144, 1663–1672.

Lindenmayer DB, Blanchard W, McBurney L, Blair D, Banks S, Liken GE, Franklin JF, Laurance WF, Stein J & Gibbons P (2012) Interacting factors driving a major loss of large trees with cavities in a forest ecosystem. *PLOS One* 7, e41864.

Lindenmayer DB, Blanchard W, McBurney L, Blair D, Driscoll D, Smith AL & Gill AM (2013) Fire severity and landscape context effects on arboreal marsupials. *Biological Conservation* 167, 137–148.

Lindenmayer DB, Blanchard W, Blair D, McBurney L & Banks SC (2017a) Relationships between tree size and occupancy by cavity-dependent arboreal marsupials. *Forest Ecology and Management* 391, 221–229.

Lindenmayer DB, Cunningham RB & Donnelly CF (2017b) Decay and Collapse of Trees with Hollows in Eastern Australian Forests: Impacts on Arboreal Marsupials. *Ecological Applications* 7, 625–641.

Lindenmayer DB, Blanchard W, Blair D, McBurney L, Stein J & Banks SC (2018a) Empirical relationships between tree fall and landscape-level amounts of logging and fire. *PLoS One* 13, e0193132.

Lindenmayer DB, Wood J, MacGregor C, Foster C, Scheele B, Tulloch A & O'Loughlin LS (2018b) Conservation conundrums and the challenges of managing unexplained declines of multiple species. *Biological Conservation* 221, 279–292.

Lindenmayer DB, Blanchard W, Blair D, McBurney L, Taylor C, Scheele BC, Westgate MJ, Robinson N & Foster C (2020) The response of arboreal marsupials to long-term changes in forest disturbance. *Animal Conservation.* Available on the internet at: https://doi.org/10.1111/acv.12634

Lindenmayer D, Blair D, McBurney L, Banks S & Bowd E. (2021) Ten years on – a decade of intensive biodiversity research after the 2009 Black Saturday wildfires in Victoria's Mountain Ash forest. *Australian Zoologist* 41, 220–230.

Lumsden LF, Nelson JL, Todd CR, Scroggie MP, McNabb EG, Raadik TA, Smith SJ, Acevedo S, Cheers G, Jemison ML & Nico MD (2013) *A New Strategic Approach to Biodiversity Management – Research Component. Arthur Rylah Institute for Environmental Research.* Unpublished Client Report for the Department of Environment and Primary Industries, Heidelberg, Victoria.

Lunney D (1987) Effects of logging, fire and drought on possums and gliders in the coastal forests near Bega, N.S.W. *Australian Wildlife Research* 13, 67–92.

Lunney D, Triggs B, Eby P & Ashby E (1990) Analysis of scats of dogs *Canis familiar* and foxes *Vulpes vulpes* (Canidae: Carnivora) in coastal forests near Bega New South Wales. *Australian Wildlife Research* 17, 61–68.

Lunney D, Menkhorst P, Winter J, Ellis M, Strahan R, Oakwood M, Burnett S, Denny M & Martin R (2008) *Petauroides volans*. In 'IUCN red list of threatened species.' Version 2012.2. Viewed: 11 December 2012. Available on the internet at: http://www.iucnredlist.org/species/40579/166500472

Macfarlane MA (1988) Mammal populations in Mountain Ash forests (*Eucalyptus regnans*) forests of various ages in the Central Highlands of Victoria. *Australian Forestry* 51, 14–27.

Mackowski DM (1984) The otengeny of hollows in Blackbutt (*Eucalyptus pilularis*) and its relevance to the management of forests for possums, gliders and timber, in AP Smith & ID Hume (eds) *Possums and Gliders*. Surrey Beatty and Sons, Chipping Norton. pp. 553–662.

Maloney KS (2007) The status of the greater glider *Petauroides volans* in the Illawarra region. M.Sc.-Res. Thesis, School of Biological Sciences, University of Wollongong, Wollongong.

Matusick G, Ruthrof KK, Brouwers NC, Dell B & Hardy GE (2013) Sudden forest canopy collapse corresponding with extreme drought and heat in a mediterranean-type eucalypt forest in southwestern Australia. *European Journal of Forest Research* 132, 497–510.

McCarthy MA & Lindenmayer DB (1999a) Conservation of the greater glider (*Petauroides volans*) in remnant native vegetation within exotic plantation forest. *Animal Conservation* 2, 203–209.

McCarthy MA, & Lindenmayer DB (1999b) Incorporating metapopulations dynamics of greater gliders into reserve design in disturbed landscapes. *Ecology* 80, 651–667.

McGregor DC, Padovan A, Georges A, Krockenberger A Yoon H & Youngentob KN (2020) Genetic evidence supports three previously described species of greater glider, *Petauroides volans, P. minor, and P. armillatus. Scientific Reports* 10, 19284. Available on the internet at: https://doi.org/10.1038/s41598-020-76364-z

McKiernan AB, Hovenden MJ, Brodribb TJ, Potts BM, Davies NW & O'Reilly-Wapstra JM (2014) Effect of limited water availability on foliar plant secondary metabolites of two Eucalyptus species. *Environmental and Experimental Botany* 105, 55–64.

McLean CM, Kavanagh RP, Penman T & Bradstock R (2018) The threatened status of the hollow dependent arboreal marsupial, the greater glider (*Petauroides volans*), can be explained by impacts from wildfire and selective logging. *Forest Ecology and Management* 415-16, 19–25.

McKay GM (1989) Petauridae. In DW Walton & BJ Richardson (eds) *Fauna of Australia. Vol. 1B. Mammalia.* Australian Government Printing Service. Canberra. pp. 665–679.

McKay GM (2008) Greater glider *Petauroides volans*, in S Van Dyck & R Strahan (eds) *The Mammals of Australia*. Third edition. Reed New Holland, Sydney. pp. 240–242.

Menkhorst PW (1984) Use of nest boxes by forest vertebrates in Gippsland: acceptance, preference and demand. *Australian Wildlife Research* 11, 255–26.

Moore BD, Wallis IR, Marsh KJ & Foley WJ (2004) The role of nutrition in the conservation of the marsupial folivores of eucalypt forests, in D Lunney (ed) *Conservation of Australia's Forest Fauna*. Second edition. pp 549–575.

NSW National Parks and Wildlife Service South Coast Branch (2020) *Post-fire fauna surveys in coastal national parks of the Shoalhaven Area, NSW National Parks and Wildlife Service, South Coast Branch; Murramarang National Park, Meroo National Park, Conjola National Park and Corramy Regional Park*. Report prepared by Phillip Craven and Gary Daly.

Nelson JL, Scroggie MP, Durkin LK, Cripps JK, Ramsey DSL & Lumsden LF (2018) *Estimating the density of the greater glider in the Strathbogie Ranges, North East Victoria, with an assessment of coupes scheduled for timber harvesting in 2018.* Arthur Rylah Institute for Environmental Research Technical Report Series No. 293. Department of Environment, Land, Water and Planning, Heidelberg, Victoria.

Parliament of Victoria (2020) Research note No. 1. 2019-2020 Bushfires, Quick Guide. Parliamentary Library Services, Parliament of Victoria. Available on the internet at:

https://www.parliament.vic.gov.au/publications/research-papers/download/36-research-papers/13904-bushfires-2019-20

Parnaby H, Lunney D, Shannon I & Fleming M (2010) Collapse rates of hollow-bearing trees following low intensity prescription burns in the Pilliga forests, New South Wales. *Pacific Conservation Biology* 16, 209–220.

Pope ML, Lindenmayer DB & Cunningham RB (2004) Patch use by the greater glider (*Petauroides volans*) in a fragmented forest ecosystem. I. Home range size and movements. *Wildlife Research* 31, 559–568.

Possingham HP, Lindenmayer DB, Norton TW & Davies I (1994) Metapopulation viability analysis of the greater glider *Petauroides volans* in a wood production area: *Biological Conservation*, 70, 227–236.

Queensland Herbarium (2018) *Threatened species and ecosystems at SWBTA: A summary of threatened fauna, flora and ecosystems surveys conducted November 2017 – April 2018*. Department of Energy and Science, Brisbane Botanic Gardens.

Ross Y (1999) Hollow-bearing trees in native forest permanent inventory plots in southeast Queensland. *Forest Ecosystem Research and Assessment Technical Papers*, pp. 99–123. Queensland Department of Natural Resources.

Rübsamen K, Hume ID, Foley WJ & Rübsamen U (1984) Implications of the large surface area to body mass ratio on the heat balance of the greater glider *Petauroides volans*. *Journal of Comparative Physiology, B-Biochemical, Systemic, and Environmental Physiology* 154, 105−111.

Smith AP, Moore DM & Andrews SP (1994a) Fauna of the Grafton and Casino Forestry Study Areas description and assessment of forestry impacts. Report for State Forests of New South Wales. Austeco Environmental Consultants, Armidale.

Smith AP, Andrews SP, Gration G, Quinn D & Sullivan B (1994b) Description and assessment of forestry impacts on fauna of the Urunga - Coffs Harbour Forestry Management Area. Report for State Forests of New South Wales. Austeco Environmental Consultants, Armidale.

Smith GC, Mathieson M & Hogan L (2007) Home range and habitat use of a low-density population of greater glider, *Petauroides volans* (Pseudocheiridae: Marsupialia), in a hollow-limiting environment. *Wildlife Research* 34, 472–483.

Smith GC, Lewis T & Hogan L (2015) Fauna community trends during early restoration of alluvial open forest/woodland ecosystems on former agricultural land. *Restoration Ecology* 23, 787−799.

Smith P & Smith J (2018) Decline of the greater glider (*Petauroides volans*) in the lower Blue Mountains, New South Wales. *Australian Journal of Zoology* 66, 103−114.

Smith P & Smith J (2020) Future of the greater glider (*Petauroides volans*) in the Blue Mountains, New South Wales. *Proceedings of the Linnean Society of New South Wales* 142, 55−66.

Smith P & Smith J (2021) *Arboreal mammal survey of the Bigga-Tuena area*. Final report. New South Wales.

Standards Reference Group SERA (2021) *National Standards for the Practice of Ecological Restoration in Australia*. Edition 2.2. Society for Ecological Restoration Australasia. Available on the internet at: http://www.seraustralasia.com/standards/home.html

Taylor AC, Tyndale-Biscoe CH & Lindenmayer DB (2007) Unexpected persistence on habitat islands: genetic signatures reveal dispersal of a eucalypt-dependent marsupial through a hostile pine matrix. *Molecular Ecology* 16, 2655–2666.

Taylor BD & Goldingay RL (2009) Can road-crossing structures improve population viability of an urban gliding mammal? *Ecology and Society* 14, 13. Available on the internet at: https://www.ecologyandsociety.org/vol14/iss2/art13/

Taylor C, McCarthy MA & Lindenmayer DB (2014) Nonlinear effects of stand age on fire severity. *Conservation Letters* 7, 355–370.

Tyndale-Biscoe CH & Smith RFC (1969a) Studies on the marsupial glider *Schoinobates volans* (Kerr). III. Response to habitat destruction. *Journal of Animal Ecology* 38, 651–659.

Tyndale-Biscoe CH & Smith RFC (1969b) Studies on the marsupial glider, *Schoinobates volans* (Kerr). II. Population structure and regulatory mechanisms. *Journal of Animal Ecology* 38, 637–650.

Van Dyck S & Strahan R (2008) *The Mammals of Australia*. Reed New Holland, Sydney.

van der Ree R (1999) Barbed wire fencing as a hazard for wildlife. *The Victorian Naturalist* 116, 210–217.

van der Ree R, Ward SJ & Handasyde KA (2004) Distribution and conservation status of possums and gliders in Victoria, in RL Goldingay & SM Jackson (eds) *The Biology of Australian Possums and Gliders*. Surrey Beatty and Sons, Sydney. pp. 91–110.

Vinson SG, Johnson AP & Mikac KM (2020) Current estimates and vegetation preferences of an endangered population of the vulnerable greater glider at Seven Mile Beach National Park. *Austral Ecology*. Available on the internet at: https://doi.org/10.1111/aec.12979

Wagner B, Baker PJ, Stewart SB, Lumsden LF, Nelson JL, Cripps JK, Durkin LK, Scrogge M & Nitschke CR (2020) Climate change drives habitat contraction of a nocturnal arboreal marsupial at it's physiological limits. *Ecosphere* 11, e03262. Available on the internet at: https://doi.org/10.1002/ecs2.3262

Wallis RL & Brunner H (1986) Changes in mammalian prey of foxes, *Vulpes vulpes* (Carnivora: Vanidae) over 12 years in a forest park near Melbourne, Victoria. *Australian Mammology* 10, 43–44.

Woinarski JCZ, Burbidge AA & Harrison PL (2014) *The Action Plan for Australian Mammals 2012*. CSIRO Publishing, Collingwood.

Woinarski JCZ, McCosker JC, Gordon G, Lawrie B, James C, Augusteyn J, Slater L & Danvers T (2006) Monitoring change in the vertebrate fauna of central Queensland, Australia, over a period of broad-scale vegetation clearance, 1975-2002. *Wildlife Research* 33, 263–274.

Wormington K & Lamb D (1999) Tree hollow development in wet and dry sclerophyll eucalypt forest in south-east Queensland, Australia. *Australian Forestry* 62, 336–345. Available on the internet at: https://doi.org/10.1080/00049158.1999.10674801

Youngentob KN, Wallis IR, Lindenmayer DB, Wood JT, Pope ML & Foley WJ (2011) Foliage influences tree choice and landscape use of a gliding marsupial folivore. *Journal of Chemical Ecology* 37, 71–84. Available on the internet at: https://doi.org/10.1007/s10886-010-9889-9

Youngentob KN, Yoon H-J, Coggan N & Lindenmayer DB (2012) Edge effects influence competition dynamics: A case study of four sympatric arboreal marsupials. *Biological Conservation* 155, 68–76.

Youngentob KN, Wood JT & Lindenmayer DB (2013) The response of arboreal marsupials to landscape context over time: a large-scale fragmentation study revisited. *Journal of Biogeography* 40, 2082–2093. Available on the internet at: https://doi.org/10.1111/jbi.12158

Youngentob KN, Yoon H-J, Stein J, Lindenmayer DB & Held AA (2015) Where the wild things are: using remotely sensed forest productivity to assess arboreal marsupial species richness and abundance. *Diversity and Distributions* 21, 977–990.

Youngentob KN, Lindenmayer DB, Marsh KJ, Krockenberger AK & Foley WJ (2021) Food intake: an overlooked driver of climate change casualties? *Trends Ecol Evol* 36, 676–678.

Other sources cited in the advice

Armstrong KN (2021) Personal communication via email, 24 June 2021. University of Adelaide.

Atlas of Living Australia (2021). OZCAM records of *Petauroides*. Available at URL: http://ozcam.org.au/. Accessed 17 August 2021. https://doi.org/10.26197/ala.5d4b8ad1-7607-4f17-8193-861b8f5d05a3

Blake B (2020) Personal communication by email, 29 September 2020. Proconpest – Contractor for World Wide Fund for Nature (WWF).

Bluff L (2020) Personal communication by email, 15 September 2020. Acting Regional Manager; Natural Environment Programs, Gippsland; Forest, Fire and Regions. Department of Environment, Land, Water and Planning (Vic), Melbourne.

Canberra Nature Map (2019) Viewed: 21 October 2019. Available on the internet at: https://canberra.naturemapr.org/

Cobern C (2015) Personal communication by email, 9 November 2015. Landcare Coordinator, Upper Goulburn Landcare Network, Victoria.

DEHP (Department of Environment and Heritage Protection) (2015) *Submission on the eligibility of Petauroides volans (greater glider) to be categorised as Vulnerable on the EPBC Act threatened species list.* Received 25 November 2015. Department of Environment and Heritage Protection (Qld), Brisbane.

DELWP (Department of Environment, Land, Water and Planning) (2019) Personal communication by email, 15 October 2019. Department of Environment, Land, Water and Planning (Vic), Melbourne.

DELWP (Department of Environment, Land, Water and Planning) (2021) Personal communication by email, 22 April 2021. Department of Environment, Land, Water and Planning (Vic), Melbourne.

Eyre T (2021) Personal communication by email, 11 January 2021. Queensland Herbarium.

Gippsland Environment Group (2015) Personal communication by email, 24 November 2015.

Jones KE, Bielby J, Cardillo M, Fritz SA, O'Dell J, Orme CDL, Safi K, Sechrest W, Boakes EH, Carbone C, Connolly C, Cutts MJ, Foster JK, Grenyer R, Habib M, Plaster CA, Price SA, Rigby EA, Rist J, Teacher A, Bininda-Emonds ORP, Gittleman JL, Mace GM & Purvis A (2009) PanTHERIA: a species-level database of life history, ecology, and geography of extant and recently extinct mammals. *Ecology* 90, 2648.

Liepa D (2020) Personal communication via phone conversation, 10 September 2020. Greening Australia.

Moorrees A (2020) Personal communication via phone conversation, 14 September 2020. Department of Environment, Land, Water and Planning (Vic), Melbourne.

Nelson J (2021) Personal communication by email, 16 and 19 April 2021. Department of Environment, Land, Water and Planning (Vic), Melbourne.

Pacifici M, Santini L, Di Marco M, Baisero D, Francucci L, Grottolo Marasini G, Visconti P, Rondinini C (2013) Database on generation length of mammals. 5427 data records. Online at http://doi.org/10.5061/dryad.gd0m3, version 1.0 (last updated on 2013-08-27), Resource ID: 10.5061/dryad.gd0m3, Data Paper ID: doi: 10.3897/natureconservation.5.5734

Rickards P (2015) Personal communication by email, 24 November 2015. Owner of forested property in East Gippsland, Victoria.

Ridgeway P (2021) Personal communication by email, 6 January 2021. Senior land Services Officer (Biodiversity), Greater Sydney Local Land Services.

Smith J (2015) Personal communication by email, 22 November 2015. P&J Smith Ecological Consultants, New South Wales.

Smith J (2020) Personal communication by email, 10 December 2020. P&J Smith Ecological Consultants, New South Wales.

Smith P & J (2021) Personal communication by email, 24 June 2021. P&J Smith Ecological Consultants, New South Wales.

Stimson N (2021) Personal communication by email, 26 June 2021. Enviro Images, New South Wales.

Vic SAC (2015) *Submission on the EPBC Act assessment of the greater glider. Received 25 November 2015.* Victorian Scientific Advisory Committee. Department of Environment, Land, Water and Planning. State Government of Victoria, Melbourne.

VicForests (2021) *Submission on the consultation on species listing eligibility and conservation actions* – Petauroides volans *(greater glider (southern)).* Received 24 June 2021. Melbourne.

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

Attachment A: Listing Assessment for *Petauroides volans* (greater glider (southern and central))

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	100 000	Unknown	There is no robust estimate of the population size of the greater glider (southern and central). Woinarski et al. (2014) estimated over 100 000 mature individuals, and Nelson et al. (2018) estimated a subpopulation size of 69 000 in the Strathbogie ranges in Vic.
Trend	declining			Declines in occupancy of the greater glider (southern and central) have been recorded for over two decades in the Central Highlands (Lumsden et al. 2013; Lindenmayer 2020) and East Gippsland (L Bluff 2020. pers comm 15 October) regions of Vic. There have been losses of subpopulations in NSW within the Jervis Bay and Blue Mountains areas (Lindenmayer et al. 2011; Smith & Smith 2018). These declines were recorded pre-2019–20 bushfires and overall show a ≥30% decline. Post-fire surveys have indicated that in areas of high fire severity there is zero to very low occupancy (J Smith 2020. pers comm 10 December). Following the 2019–20 bushfires, an overall population decline of >20%, with local subpopulation extirpations, is estimated one year after the fires. This is expected to increase to >30% within three generations after the fires (Legge et al. 2021).

Metric	Estimate used in the assessment	Minimum plausible value	Maximum plausible value	Justification
Generation time (years)	7	6	8	The greater glider can live for 15 years (Jones et al. 2009) and reaches sexual maturity at two years of age (Tyndale-Biscoe & Smith 1969b), suggesting a generation length of six to eight years (Pacifici et al. 2013; Woinarski et al. 2014).
Extent of occurrence	752 962 km^2	752 962 km^2	1 066 146 km^2	Woinarski et al. (2014) estimated the extent of occurrence (EOO) of the greater glider (southern and central) as 752 962 km^2, calculated using records from 1993 to 2012. The 1 066 146 km^2 figure was based on the mapping of point records from 2000 to 2020, obtained from state governments, museums and CSIRO (DAWE 2021). The EOO was calculated using a minimum convex hull, based on the IUCN Red List Guidelines 2019.
Trend	contracting			The EOO has contracted since European settlement, with loss of habitat from land clearing, fragmentation, timber harvesting, inappropriate fire regimes, and climate change. Local extinctions of subpopulations have occurred recently (Lindenmayer et al. 2018b), and the EOO is likely to continue contracting due to loss of habitat from the 2019–20 bushfires and climate change.
Area of Occupancy	15 316 km^2	15 316 km^2	>20 000 km^2	The 15 316 km^2 figure is based on the mapping of point records from 2000 to 2020, obtained from state governments, museums and CSIRO (DAWE 2021). The AOO was calculated using a 2x2 km grid cell method, based on the IUCN Red List Guidelines 2019. The AOO is likely to be significantly under-estimated due to limited sampling across the species' range.
Trend	contracting			The AOO has contracted since European settlement, with loss of habitat from land clearing, fragmentation, timber harvesting, inappropriate fire regimes, and climate change. Local extinctions of subpopulations have occurred recently (Lindenmayer et al. 2018b, Smith & Smith 2020), and the AOO is likely to continue contracting due to loss of habitat from the 2019-20 bushfires and climate change.
Number of subpopulations	Unknown	Unknown	Unknown	The species has a broad distribution. The number of subpopulations is not able to be estimated due to insufficient sampling across its range.
Trend	declining			The number of greater gliders (southern and central) have been declining across its range, and together with the contracting AOO and EOO, the number of subpopulations is likely to be declining.

Metric	Estimate used in the assessment	Minimum plausible value	Maximum plausible value	Justification
Basis of assessment of subpopulation number	The greater glider (southern and central) number of subpopulations is unknown, as there is limited sampling across its broad range.			
No. locations	unknown	unknown	>10	The term 'location' defines a geographically or ecologically distinct area in which a single threatening event can rapidly affect all individuals of the taxon present (IUCN Standards and Petitions Committee 2019). There is no robust estimate of the number of locations. The 2019–20 bushfires burnt a large area of south-eastern Australia, overlapping c. 40% of the greater glider (southern and central) distribution. However, the fire intensity was highly spatially variable, with greater gliders (southern and central) persisting in at least some areas burnt at low or moderate intensity (J Smith 2020. pers comm 10 December; J Nelson 2021. pers comm 16 April). Impacts were also spatially variable, with some individuals persisting in areas burnt at high intensity, possibly due to the proximity of unburnt or low intensity burnt areas (Kavanagh et al. 2021). Thus, the number of locations may be significantly greater than 10.
Trend	declining			Climate change is likely to increase the extent, intensity and frequency of bushfires, and thus the number of locations is likely to decrease.
Basis of assessment of location number	Although the 2019-20 bushfires were extensive the habitat and landscape topography, along with the stochastic variation in fire spread, leaves numerous unburnt habitat fragments from which subpopulations may recover.			
Fragmentation	Not severely fragmented – less than 50% of the AOO are in habitat patches that cannot support minimum viable subpopulations.			
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 A2abc+4bc for listing as Endangered

The greater glider (southern and central) has a generation length of six to eight years (see Table 3). In this assessment a generation length of seven years is used, which gives a timeframe of 21 years for this criterion.

There are no robust estimates of population size or population trends of the greater glider (southern and central) across its distribution. However, declines in numbers, occupancy rates and extent of habitat have been recorded at many sites (see below). Although there are a few sites where subpopulations appear to be stable or increasing, the overall trend is one of decline.

Prior to 2019-20 bushfires

Victoria

The most comprehensive long-term monitoring program for the greater glider (southern and central) is in the *Eucalyptus regnans* (Mountain Ash) forests of the Central Highlands in Vic, where 160 permanent 1 ha sites across a 1,800 km^2 study area (in both conservation reserves and production forests, and spanning a broad range of forest ages and environmental settings) (Lindenmayer 2009) have been monitored annually since 1997. Over the period 1997–2010, the greater glider (southern and central) declined by an average of 8.8 percent per year (Lindenmayer et al. 2011) – a rate that if extrapolated over the 21-year period relevant to this assessment is 85 percent. The trend could in part be explained by lower-than-average rainfall and major bushfires, with the species not detected in any of the sites burned in 2009. However, the probability of observing the species was also significantly higher on sites located in the Yarra Ranges National Park than in forests broadly designated for pulp and timber production, and there was a significant positive relationship between the species' abundance and both the age of the forest and number of trees with hollows on a site (Lindenmayer 2009). Populations of large hollow-bearing trees in the Central Highlands are in rapid decline, with the rate of loss greatly exceeding the rate of recruitment (Lindenmayer et al. 2017a,b).

Other surveys undertaken in the Central Highlands, in both Mountain Ash and mixed species forests, indicate a significant decline in occupancy rates of the greater glider (southern and central) over the past two decades (Lindenmayer et al. 2011; Lindenmayer & Sato 2018; Lumsden et al. 2013).

In 2018, a broad-scale survey of 80 sites (500 m off-track transects) spread across central and north-eastern Vic found low numbers of greater gliders (southern and central) at the majority of sites. Despite many of the sites supporting seemingly suitable habitat, the species was detected on fewer than half (41 percent) of the transects. On average, 0.93 gliders (range 0–6) were detected per 500 m transect (DELWP unpublished data). Surveys in 2019 conducted at 63 sites within eastern Vic also found low numbers of the species, with individuals detected on only 19 percent of sites (0.21 gliders/500 m transect, range 0–2). Based on records held in the Victorian Biodiversity Atlas and anecdotally, these results suggest the species has declined across this area (DELWP 2019. pers comm 15 October).

In contrast, surveys using the same broad-scale survey methodology in the Strathbogie Ranges in north-eastern Vic found relatively high densities of gliders, with 4.92 gliders detected on average per transect (range 0–14; Nelson et al. 2018). Analyses of the survey data estimated the number of greater gliders (southern and central) within the Strathbogie Ranges to be 69 000, although with relatively broad confidence intervals (95 percent confidence interval 3000–121 000 individuals). A comparison of data from three surveys conducted in the Strathbogie Ranges in 1983 (Land Conservation Council 1984), 1997 (Downes et al. 1997) and 2017 (Nelson et al. 2018), suggests that the subpopulation in the Strathbogie Ranges has not declined over a 34 year period to the extent that has been observed elsewhere in Vic (Nelson et al. 2018).

Major bushfires in 2003, 2006−2007 and 2009 burnt large areas of the greater glider (southern and central) range in Vic, and further fragmented its distribution as evidenced by surveys and species records (Lumsden et al. 2013; Vic SAC 2015). Following the 2009 bushfires, 79 percent of large living trees with cavities died in the Mountain Ash forests, with no recruitment of new large cavity-bearing trees by 2011 (Lindenmayer et al. 2013). The abundance of greater gliders (southern and central) declined at burned sites, as well as at unburnt sites that were surrounded by burned forest (Lindenmayer et al. 2013). Reoccupation of burnt sites in subsequent years is a slow process due to the small home ranges (1−4 ha) of the species and its limited dispersal capabilities (L Lumsden pers comm, cited in Vic SAC 2015). It also depends on there being no further significant fires in the interim (Vic SAC 2015). Since the 2009 fires, which burnt the Kinglake East Bushland Reserve and nearby areas, spotlighting records of greater gliders (southern and central) in these areas have significantly declined (C Cobern 2015. pers comm 9 November). The occupancy model in Lumsden et al. (2013) predicts that areas most likely to be occupied following the 2009 fires are now patchily distributed.

However, evidence of declines in occupancy in some unburnt sites in the same parts of Vic (Lumsden et al. 2013) suggest that factors other than fire are involved in the species' decline (Vic SAC 2015). A decline in suitable browse due to water stress is probably a contributing factor, as central Vic was significantly hotter and drier than normal during 2001−2009 (Vic SAC 2015). Occupancy modelling by Lumsden et al. (2013) and Wagner et al. (2020) shows that the degree of site occupancy is positively associated with site ruggedness, vegetation lushness and terrain wetness.

In East Gippsland, analysis of results from a survey of 107 sites, comprising 49 sites with previous records of greater gliders (southern and central) and 58 randomly stratified sites, found a decline in occupancy rates of about 50 percent compared to about 20 years ago (L Bluff 2020. pers comm 15 October). The survey was undertaken in 2015 and results were compared to the pre-logging survey period 1988-1995. Although the occupancy rate of all arboreal mammals that were detected in sufficient numbers to enable analysis had declined across the two decades, the greater glider (southern and central) had declined more than other species. The decline in the rate of detection was highest in coastal and foothill forests, while detection rates were high only in wet and damp tableland forest on the Errinundra Plateau and Coast Range.

In the Mount Alfred State Forest, roadside spotlighting on the same route over a 30-year period used to record frequent sightings (10−15 animals on each occasion), but only a single greater glider (southern and central) was sighted in the 18 months leading up to November 2015 (Gippsland Environment Group 2015 pers comm 24 November).

New South Wales and the Australian Capital Territory

At Jervis Bay in Booderee National Park, 110 permanent 1 ha sites (stratified across vegetation types and fire histories) were established in 2002. Lindenmayer et al. (2011) reported a highly significant decline of greater gliders (southern and central), from the species being present in 22 of the sites in 2002, to absence from all sites since 2007. The greater glider (southern and central) has not been recorded in the National Park since 2006 and appears to have been extirpated from the area, for reasons unclear (Lindenmayer et al. 2018b).

At Murphy's Glen in the Blue Mountains, spotlighting undertaken between 1986 and 2014 shows that the species used to be consistently and regularly detected, but by 2010 was difficult to detect and likely no longer present (J Smith 2015. pers comm 22 November). However, spotlighting undertaken in 2015 recorded greater gliders (southern and central) on each of the three occasions (1, 2 and 5 individuals), which indicates that numbers may be recovering (J Smith 2015. pers comm 22 November). Anecdotal reports, including from local ecologists, indicated similar declines elsewhere in the lower Blue Mountains, and the NSW Bionet Atlas confirms a marked drop in records in the region (Blue Mountains National Park: 357 records 1990–2004, eight records 2004–2014. Blue Mountains LGA: 142 records 1990–2004, one record 2004–2014, five records 2018–2020 and only one record for 2020) (J Smith 2015. pers comm 22 November). The decline of the greater glider (southern and central) in the lower Blue Mountains is mostly likely due to the effects of increased temperatures as a result of climate change (Smith & Smith 2018, 2020). An autopsy undertaken in January 2020 on two individuals (which were found walking on the ground in the daytime), reported that they had both died from drought and extreme heat events (i.e. heat stress and dehydration) (P Ridgeway 2021. pers comm 6 January).

An isolated subpopulation at Royal National Park was thought to be lost due to fire and regional-scale decline in the Illawarra area. Following the 1994 bushfire, which burnt more than 90 percent of the park, the first confirmed sighting of a greater glider (southern and central) in Royal National Park was in 2012 (Andrew et al. 2014), although a number of surveys had been conducted since 1994 (Andrew 2001; Maloney 2007; Andrew et al. 2014).

Near Bombala in southern NSW, Kavanagh and Webb (1998) monitored greater gliders (southern and central) in 500 ha of wood production forest, and found that the subpopulation declined in all timber harvesting compartments and had not recovered eight years after harvesting. However, the effects of logging were compounded by the independent effects of predation by powerful owls, and the overall declines of greater glider in this study were attributed to predation (Kavanagh 1988).

About 30 years after clearing of eucalypt forests in Tumut, Lindenmayer et al. (1999) found that the occupancy rate of greater gliders (southern and central) in remnant patches was still lower (21 percent) compared to that in surrounding forest (38 percent), indicating that recolonization following clearing occurs slowly. It is unclear, following such disturbances, whether subpopulations recover to their former levels or persist at lower levels.

In the Dorrigo, Guy Fawkes and Chaelundi Plateaux of north-eastern NSW, surveys for the greater glider (southern and central) at 30 sites in wet sclerophyll forest recorded a density of 27.6 individuals per km, in unlogged forest with no fire history (McLean et al. 2018).

Queensland

In central Qld, the abundance of greater gliders (southern and central) declined by 89 percent across a series of 31 woodland sites sampled initially in 1973−76 and re-sampled in 2001−02 (Woinarski et al. 2006). The species is continuing to decline, based on anecdotal observations over a 20-year period (DEHP 2015) and evidence of a decline in large, hollow-bearing trees due to past timber harvesting activities and repeat prescribed burning (Eyre 2005; Eyre et al. 2010). There has been a decline in living hollow-bearing trees (25 percent) and stags (40 percent) over a 20-year period (1998–2018) in the St. Marys State forest area (T Eyre 2021 pers comm 11 January). Once habitat trees are lost from the system, the length of time required for the development/recruitment of replacement habitat trees appropriate for the species is largely prohibitive (Smith et al. 2015).

After the 2019-20 bushfires

The full impact of the 2019-20 bushfires on the greater glider (southern and central) has yet to be determined but the population is likely greatly reduced. The fires may have accelerated any ongoing population decline, with approximately 40 percent of the species' distribution overlapping with the fire-affected areas (Legge et al. 2021). These fires covered an unusually large area and, in many places, burnt with an unusually high intensity. Its pre-fire imperilment, together with the extent of mortality as a result of fire and the unfavourable post-fire conditions (loss of hollows, increased susceptibility to predators, and loss of food resources), as well as a reduction in future recruitment, led to the greater glider (southern and central) being identified as one of the highest priority species for urgent management intervention by the Wildlife and Threatened Species Bushfire Recovery Expert Panel (Legge et al. 2020).

It is known that the greater glider (southern and central) is highly susceptible to fire events, with population declines of 50 percent documented in some areas (McLean et al. 2018) and extirpation with slow recovery documented in others (Andrew et al. 2014). Following the 2019-20 bushfires, on-ground surveys in some areas are still to be conducted, and baseline data are missing on population size, distribution and density throughout the range of the species. The majority of records are from the eastern areas of NSW and Vic, which were extensively burnt (DPIE 2020; Parliament of Victoria 2020). Post-fire field survey data available to date are summarised in the section below.

In addition to direct observations (see below), an expert elicitation exercise has been run to estimate the likely decline in greater glider (southern and central) populations due to fires of varying intensity (Legge et al. 2021). This was then combined with a GIS analysis of overlap of the distribution of the greater glider (southern and central) with the fire footprint to provide an overall estimate of the likely population decline due to the fires. The result was an estimated loss of 24 percent of the population (range 17–31%) one year after the fires, assuming current management conditions (Legge et al. 2021). This estimate rises to 33 percent (range 18—48%) three generations after the fires.

Victoria

Surveys currently underway (April 2021) are focused predominantly on lightly burnt and unburnt habitat within the fire ground, but also some areas burnt at moderate to higher severity (DELWP 2021. pers comm 22 April). Surveys have been designed to visit pre-fire records of the greater glider (southern and central) near Swifts Creek and Bendoc in East Gippsland. Interim results for surveys along 500 m transects at 11 sites (one third of all sites planned for surveys) have detected the species at four lightly burnt sites, as well as at two sites that were burnt at higher severity; compared to pre-fire records, the numbers of individuals detected were lower and the species was not detected at five sites where they were previously recorded (J Nelson 2021. pers comm 19 April). Surveys at 30 sites in lower elevation forests in East Gippsland (from Cabbage Tree Creek to Drummer State Forest), that burnt at low severity, did not detect any individuals (DELWP 2021. pers comm 22 April).

Greening Australia recorded nest boxes being utilised by greater gliders (southern and central) post-fire in East Gippsland (D Liepa 2020. pers comm 10 September), and spotlighting surveys (500 m transects at 24 sites) recorded the species in low numbers at some sites. Individuals were detected at seven of the 18 sites where they were previously recorded, suggesting a 60 percent decline due to the fires (B Blake 2020. pers comm 25 September). A further spotlighting survey of 500 m transects undertaken in Mallacoota, Far East Gippsland, detected the species in only one of 12 transects where they were recorded previously, indicating a 90 percent decline (Burns & Atkins 2021). The one detection site had low canopy scorching.

Limited spotlighting surveys undertaken in the Tallarook Range in the Central Highlands, from October 2020 to March 2021, recorded the species within an area of less than 10 km^2 (N Stimson 2021. pers comm 24 June). This subpopulation may be geographically isolated and restricted to the central area of the Tallarook Range plateau, however further survey work is required to determine this.

New South Wales

South Coast

Spotlighting surveys at 71 sites, undertaken at Murramarang, Meroo and Conjola National Parks, and Corramy Regional Park in May and June of 2020, reported on average a 70 percent decline in the numbers of greater gliders (southern and central) detected at these sites, compared to surveys undertaken prior to the 2019-20 bushfires (NSW NPWS 2020).

Two post-fire surveys were undertaken in the southern tablelands east of Bombala, in November 2020 and April-May 2021 respectively. The sites were distributed across elevations ranging 800–1100 m a.s.l. A total of 18 spotlighting sites/transects (each 1000 m) were surveyed using similar methods to previous surveys undertaken in the area, with sites stratified according to modelled fire severity classes in 2019-20. Greater gliders (southern and central) were previously recorded at all 18 transects on almost every sampling occasion; in 2020-21 the species was still present at all sites but in greatly reduced numbers on the burnt sites. A negative relationship was found between the species' abundance and increasing fire severity in the local landscape, and the number of fires over the past 30 years was also found to be negatively associated with the species' abundance (Kavanagh et al. 2021).

Blue Mountains Region

In the Blue Mountains area, sites with greater glider data prior to the 2017-19 drought and 2019-20 fires were re-surveyed during 2020-21. The surveys involved three one-hour spotlight searches of sixteen 500m transects that previously supported the species, comprising eight burnt and eight unburnt transects. In the burnt transects, no greater gliders (southern and central) were detected at the two sites which had total canopy loss, whereas they were detected at reduced numbers at the six transects which had 44–77% canopy loss. The overall result was a 36% decline (p=0.00012) in the mean detection density for the six burnt transects between 2015-18 and 2020-21. However, in the eight unburnt transects there was also a reduction in numbers, with a decline (p=0.014) of 51% between 2015-16 and 2021-21 (P & J Smith 2021, pers comm 24 June).

It is estimated that 84% of known greater glider (southern and central) habitat in the Greater Blue Mountains World Heritage Area (GBMWHA) was burnt in the 2019-20 fires, with 50% burnt at low-moderate severity and 34% burnt at high to extreme severity (P & J Smith 2021. pers comm 24 June). This equates to an estimated overall decline of 60% in the subpopulation as a result of the drought, heatwaves and bushfires of 2019-20. This estimate is preliminary, with further surveys planned later in 2021 (P & J Smith 2021. pers comm 24 June).

Crookwell

Using the same methodology as for the Blue Mountains, P & J Smith (2021) surveyed greater gliders (southern and central) in five transects in reserves in the Bigga-Tuena area north-west of Crookwell. The transects were surveyed in both spring 2020 and autumn 2021. The area was unaffected by the 2019-20 bushfires but had experienced the severe drought and heatwaves of 2019. They found that numbers on the three transects where the species was previously recorded declined by 43% (p= 0.014) between 2017-18 and 2020-21. They also found that numbers in the five transects declined by 53% (p=0.006) between spring 2020 and autumn 2021. The reason for the latter decline is unclear. It may be the result of predation by powerful owls, which were recorded on four of the five transects, or long-term physiological impacts from the extreme conditions the gliders endured in 2019.

Far North Coast

Two post-fire surveys were undertaken between Coffs Harbour, Dorrigo, Glen Innes and Grafton, in November 2020 and April-May 2021 respectively. The sites were distributed across elevations ranging 30–1330 m a.s.l. A total of 94 spotlighting sites/transects (each 500 m) were surveyed using similar methods to previous surveys undertaken in the area, with sites stratified according to modelled fire severity classes in 2019-20. Greater gliders (southern and central) were recorded at 57 of the 75 sites where they had been recorded previously (76%), and at an additional 3 sites where they had not been recorded previously. Abundance remained similar in many areas after the 2019-20 bushfires, particularly in the higher elevation sites. There was only a slight negative relationship between the species' abundance and increasing fire severity in the local landscape. Many severely burnt areas supported relatively high populations while other similarly burnt areas did not, which may be due to patchiness in fire severity and the proximity of unburnt or low-severity burnt areas nearby. The number of fires over the past 30 years was also found to be negatively associated with the species' abundance (Kavanagh et al. 2021).

Queensland

Major bushfires in 2019-20 burnt part (approximately 10 percent) of the greater glider (southern and central) range in southern Queensland. While there has been no post-fire survey work undertaken for this species in Queensland to date, these fires would have caused direct and indirect mortalities through habitat loss and fragmentation, with a consequent decline in abundance of the species.

Overall population decline

The greater glider (including *P. minor)* was assessed in 2016, with the species found to be eligible for listing as Vulnerable against this criterion as follows (DoEE 2016a):

> 'There is little other published information on population trends over the period relevant to this assessment (around 21–24 years), and the above sites are not necessarily representative of trends across the species' range. However, they provide sufficient evidence to infer that the overall rate of population decline exceeds 30 percent over a 21–24-year (three generation) period (Woinarski et al. 2014), and indeed may far exceed 30 percent. The population of the greater glider is thought to be declining due to habitat loss, fragmentation, extensive fire and some forestry practices, and this decline is likely to be exacerbated by climate change (Kearney et al. 2010). The species is particularly susceptible to threats because of its slow life history characteristics, specialist requirements for large tree hollows (and hence mature forests), and relatively specialised dietary requirements Woinarski et al. 2014).'

Since that determination, there is no evidence that any of the major threats to this species have substantially reduced, and the effects of climate change are likely worsening (Smith & Smith 2020; Wagner et al. 2020). The effects of the 2019–20 bushfires are in addition to ongoing declines.

Overall decline can be estimated by combining the ongoing decline of 30 percent (see above) with decline due to the 2019–20 bushfires, i.e. *Past decline + Decline due to fires* Population proportion remaining after past decline*. Using decline rates of 24 percent (range 17–31%) one year after the fires and 33 percent (range 18–48%) three generations after the fires, as determined by Legge et al. (2021), gives an overall decline over the past three generations (21 years) of 47 percent (Criterion 1A2) and an overall decline over a three generation period including both the past and the future of 53 percent (Criterion 1A4). However, large-scale fire and catastrophic drought were not accounted for during projections of future declines (Legge et al. 2021). Given that Australia is predicted to continue to experience increased frequency, intensity and scale of bushfires into the future (BOM & CSIRO 2020), declines over a period including both the past and the future may be even greater.

Conclusion

Given the uncertainty in the estimates of overall decline, the Committee considers that the species has undergone a severe reduction in numbers of at least 50 percent over the past three generation period (21 years for this assessment) (Criterion 1A2), and over a three generation period that includes both the past and the future (Criterion 1A4). The reduction has not ceased and the cause has not ceased. Therefore, the species has met the relevant elements of Criterion 1 to make it eligible for listing as Endangered.

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 subpopulations; (v) number of mature individuals			
(c)	Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations; (iv) number of mature individuals			

Criterion 2 evidence

Not eligible

The Extent of Occurrence (EOO) for the greater glider (southern and central) is estimated at 1 066 146 km^2, and the Area of Occupancy (AOO) estimated at 15 316 km^2. These figures are based on the mapping of point records from 2000 to 2020, obtained from state governments, museums and CSIRO (DAWE 2021). 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 2019. Woinarski et al. (2014) noted that the AOO, which they estimated to be 15 244 km^2, and the EOO which they estimated to be 752 962 km^2, are likely to be significant underestimates due to limited sampling across the occupied range of the greater glider (southern and central).

Following assessment of the data the Committee considers that the species is not eligible for listing in any category under this criterion as neither the EOO or AOO are 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 subpopulation	**≤ 50**	**≤ 250**	**≤ 1,000**
(a) (ii) % of mature individuals in one subpopulation =	**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, but available estimates suggest that the number of mature individuals is substantially greater than 10 000. Lunney et al. (2008) considered that the greater glider (both southern and northern) had a 'presumed large population' and was 'locally common'. In NSW, Kavanagh (2004) considered it 'widespread and common… particularly in north-eastern NSW'. Density estimates in Vic range from 0.6 to 2.8 individuals per hectare (Henry 1984; van der Ree et al. 2004; Nelson et al. 2018), and across its broader distribution density ranges from 0.01 to 5 individuals per hectare (Kavanagh 1984; Kehl & Borsboom 1984; Smith & Smith 2018; Vinson et al. 2020). However, it is noted that some of these estimates were made prior to recent population declines.

Woinarski et al. (2014) estimated the number of mature individuals to be greater than 100 000. Using a mark-recapture distance sampling approach during surveys of the Strathbogie Ranges in Vic in 2017, the subpopulation in this 21 200 ha area alone was estimated to have 69 000 individuals (Nelson et al. 2018). The Vic Government estimates that approximately 32 percent of the greater glider (southern and central) modelled range within the state was within the fire footprint, and 16 percent was burnt at high intensity. Thus, it is unlikely that the population of greater glider (southern and central) has been reduced to below 100 000 mature individuals.

Following assessment of the data the Committee considers that the species is not eligible for listing in any category under this criterion as the total population size is not 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 km² 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

Woinarski et al. (2014) estimate the population size to be greater than 100 000 mature individuals (see Criterion 3) and it is highly unlikely that the number of mature individuals is less than 1000. Additionally, the greater glider (southern and central) does not meet the quantitative threshold for Vulnerable under sub-criterion D2. The area of occupancy (AOO) is estimated to be 15 532 km² and the species occurs at more than five locations.

Following assessment of the data the Committee considers that the species is not eligible for listing in any category under this criterion as the number of mature individuals is not low.

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

Several local-level population viability analyses have been undertaken – e.g. for Yarra State Forest Vic (Possingham et al. 1994), Tumut NSW (Lindenmayer et al. 2001), Brisbane Qld (Taylor & Goldingay 2009) – but none for the full species (Woinarski et al. 2014).

Population viability analysis has not been undertaken. Therefore, there is insufficient information to determine the eligibility of the species 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 **transferring** *Petauroides volans* from the Vulnerable category to the Endangered category

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

Cataloguing data

This publication (and any material sourced from it) should be attributed as: Department of Climate Change, Energy, the Environment and Water 2022, *Conservation advice for Petauroides volans (greater glider (southern and central))*, Canberra.

This publication is available at the SPRAT profile for *Petauroides volans (greater glider (southern and central)).*

Department of Climate Change, Energy, the Environment and Water
GPO Box 858, Canberra ACT 2601
Telephone 1800 900 090
Web dcceew.gov.au

Version history table

Document type	Title	Date
Conservation Advice (including listing assessment	Conservation Advice for *Petauroides volans* (greater glider (southern and central))	Approved 05/07/2022
Conservation Advice (including listing assessment)	Conservation Advice for *Petauroides volans* (greater glider)	Approved 25/05/2016

The Minister approved this conservation advice and included this species in the Vulnerable category, effective from 5 May 2016

Conservation Advice

Charadrius leschenaultii

Greater sand plover

Taxonomy

Conventionally accepted as *Charadrius leschenaultii* Lesson, 1826. Charadriidae.

Other common names include: large sand plover; great, large or large-billed dotterel or sand-dotterel; Geoffroy's plover (Marchant & Higgins 1993).

The greater sand plover is a conventionally accepted species (Marchant & Higgins 1993; Christidis & Boles 2008). There are three subspecies:

- nominate subspecies *C. l. leschenaultii* which breeds in the northern parts of the Gobi Desert in Mongolia, in north-western China and southern Siberia, and spends the non-breeding season in Australasia, south-east Asia and the Indian subcontinent;
- *C. l. columbinus* which breeds in the Middle East, Turkey to southern Afghanistan, and spends the non-breeding season in the Red Sea, Gulf of Aden and the south-eastern shores of the Mediterranean Sea (Marchant & Higgins 1993); and,
- *C. l. scythicus* which breeds from Turkmenistan through south Kazakhstan and spends the non-breeding season along the coasts of eastern and southern Africa (Gill & Donsker 2015).

Note that *C. l. scythicus* was previously known as *C. l. crassirostris* until it was established that this name is pre-occupied by another plover, a subspecies of Wilson's Plover, *C. wilsonia crassirostris* (Carlos et al. 2012; Gill & Donsker 2015).

Summary of assessment

Conservation status

Vulnerable: Criterion 1 A2 (a)

The highest category for which *Charadrius leschenaultii* is eligible to be listed is Vulnerable.

Charadrius leschenaultii has been found to be eligible for listing under the following listing categories
Criterion 1: A2 (a): Vulnerable
Criterion 2: Not eligible
Criterion 3: Not eligible
Criterion 4: Not eligible
Criterion 5: Not eligible

Species can be listed as threatened under state and territory legislation. For information on the listing status of this species under relevant state or territory legislation, see http://www.environment.gov.au/cgi-bin/sprat/public/sprat.pl

Reason for conservation assessment by the Threatened Species Scientific Committee

This advice assessment of new information provided to the Committee to list *Charadrius leschenaultia.*

Public Consultation
Notice of the proposed amendment and a consultation document was made available for public comment for 47 business days between 1 October and 4 December 2015. Any comments received that were relevant to the survival of the species were considered by the Committee as part of the assessment process.

Species/Sub-species Information

Description
The greater sand plover is a small-to-medium sized shorebird (length 22–25 cm; body mass 75–100 g) with a straight longish bill that bulges towards the end but has a pointed tip. The legs are long and olive-grey (Marchant & Higgins 1993; Ward 2012).

In non-breeding plumage, the head, nape and upperparts are grey-brown and there are large grey-brown patches on the sides of the breast. The forehead eyebrow, chin, neck and underparts are white. Sexes are non-distinguishable from each other when in non-breeding plumage. However, sexes differ when in breeding plumage with males having a chestnut breast-band and rufous tinging to the head and nape and with black on the face (Marchant & Higgins 1993; Ward 2012). Juvenile birds appear similar to non-breeding adults, but the feathers of the upperparts have narrow buff fringes and indistinct dark streaking and sub-terminal bands. Juveniles may also have a buff tinge to the face, and grey-brown patches at the sides of the breast, which may extend as a wash across the breast (Marchant & Higgins 1993).

When in Australia the species is usually in non-breeding plumage and is often difficult to distinguish from the similar lesser sand plover *C. mongolus* although the greater sand plover is distinctly larger (Marchant & Higgins 1993). To untrained observers, greater sand plovers may be difficult to detect in mixed flocks of shorebirds although, when roosting, the greater sand plover tends to roost higher up the beach than other shorebirds and is usually segregated from lesser sand plovers (Marchant & Higgins 1993). Similar to the oriental plover *C. veredus*, although the greater sand plover has a smaller head, longer neck and longer wings (Marchant & Higgins 1993).

Distribution

Australian distribution

The greater sand plover breeds in the northern hemisphere and undertakes annual migrations to and from southern feeding grounds for the austral summer. The subspecies *C. l. leschenaultii* occurs in the East Asian-Australasian Flyway, EAAF (Bamford et al. 2008). Nearly three quarters of the EAAF population is in Australia during the non-breeding period (Bamford et al. 2008).

The greater sand plover distribution in Australia during the non-breeding season is widespread, although the most are found in northern Australia (Minton et al. 2006; Garnett et al. 2011; Ward 2012). In general, the distribution of this species is:

Western Australia - especially widespread between North West Cape and Roebuck Bay and also occasionally recorded along the coast of southern Western Australia;

Northern Territory - recorded from most of the coastline with the most significant areas around the Joseph Bonaparte Gulf, the coast from Anson Bay to Murgenella Creek (including the south coast of the Tiwi Islands), the northern Arnhem coast, and the Port McArthur area;

Queensland - south-eastern parts of the Gulf of Carpentaria and widespread from the Torres Strait along the eastern coast of Queensland;

New South Wales - found from the Queensland border along the coast to the Northern Rivers region with occasional records south to about Shoalhaven Heads;

Victoria - mostly recorded from Corner Inlet, Western Port and Port Phillip Bay;

Tasmania - small numbers occur in most years; and,

South Australia - mostly recorded from the Coorong, Gulf St Vincent and Spencer Gulf, as well as on the Eyre Peninsula, west to about Streaky Bay (Marchant & Higgins 1993; Barrett et al. 2003; Chatto 2003; Minton et al. 2006; Garnett et al. 2011).

This species has also been recorded on Ashmore Reef, Cocos (Keeling) Islands, Christmas Island and Lord Howe Island (Marchant & Higgins 1993).

Global distribution

The greater sand plover has an extremely large global range with the extent of occurrence estimated to be 3,460,000 km^2 (BirdLife International 2015).

The greater sand plover is one of 35 migratory shorebird species that breed in the northern hemisphere during the boreal summer and are known to annually migrate to the non-breeding grounds of Australia along the EAAF for the austral summer. In general, the EAAF stretches from breeding grounds in the Russian tundra, Mongolia and Alaska southwards through east and south-east Asia, to non-breeding areas in Indonesia, Papua New Guinea, Australia and New Zealand (Department of the Environment 2015a,b). Of the three subspecies of the greater sand plover, only *C. l. leschenaultii* occurs in the EAAF and this subspecies also occurs in the Central Asian Flyway (Bamford et al. 2008).

The greater sand plover breeds in the northern Gobi Desert of Mongolia and adjacent areas of southern Siberia; north-western China; from south-eastern Kazakhstan west to the Aral Sea and the eastern shores of the Caspian Sea, and south to Afghanistan; and at scattered sites from Azerbaijan, west into Turkey and south through Syria to Jordan (Marchant & Higgins 1993; Wiersma 1996; Gill & Donsker 2015).

The subspecies *C. l. leschenaultii* , which occurs in Australia during the non-breeding period, breeds in China, Mongolia and nearby parts of Russia (Bamford et al. 2008; Garnett et al. 2011).

Relevant Biology/Ecology

Life history

A generation time of 8 years (BirdLife International 2015) is derived from an average age at first breeding of 2 years (Cramp et al. 1983), an annual adult survival of 56% (extrapolated from congeners) and a maximum longevity of 12.6 years (Australian Bird and Bat Banding Scheme; Garnett et al. 2011).

Breeding

The migratory greater sand plover does not breed in Australia.

At breeding sites in Mongolia, north-western China and southern Siberia, the greater sand plover nests in a shallow scrape on the ground amongst sand-hills, gravel, or on other barren substrates. In these areas, this species is predominantly found in open desert or semi-arid areas that are predominantly treeless and at elevations up to 3 000 m (del Hoyo et al. 1996; BirdLife International 2015). Egg laying occurring in April and May. Clutches usually comprise three eggs (range 2-4), which are incubated by both parents for at least 24 days. The chicks fledge after about 30 days (del Hoyo et al. 1996).

General habitat

In the non-breeding grounds in Australasia, the species is almost entirely coastal, inhabiting littoral and estuarine habitats. They mainly occur on sheltered sandy, shelly or muddy beaches, large intertidal mudflats, sandbanks, salt-marshes, estuaries, coral reefs, rocky islands rock platforms, tidal lagoons and dunes near the coast (Marchant & Higgins 1993; del Hoyo et al. 1996; BirdLife International 2015).

Feeding habitat

Greater sand plovers usually feed from the surface of wet sand or mud on open intertidal flats of sheltered embayments, lagoons or estuaries (Marchant & Higgins 1993).

Roosting habitat

Greater sand plovers usually roost on sand-spits and banks on beaches or in tidal lagoons (Marchant & Higgins 1993), and occasionally on rocky points or in adjacent areas of saltmarsh (Gosper & Holmes 2002) or claypans (Collins et al. 2001). They tend to roost further up the beach than other shorebirds, sometimes well above high-tide mark (Marchant & Higgins 1993). To avoid heat stress in tropical areas, shorebirds showed a strong preference for roost sites where a damp substrate lowered the local temperature (Battley et al. 2003; Rogers et al. 2006). Approximately one day after a cyclone at Broome, Western Australia, greater sand plovers were recorded in lower than expected numbers and it was thought that some birds may have moved to sheltered areas to avoid the high winds and heavy rain associated with the cyclone (Jessop & Collins 2000).

Diet

During the breeding season, the diet of the greater sand plover consists mainly of terrestrial insects and their larvae (especially beetles, termites, midges and ants), and occasionally lizards (del Hoyo et al. 1996). During the non-breeding season, the diet mostly consists of molluscs, worms, crustaceans (especially small crabs and sometimes shrimps) and insects (including adults and larvae of termites, beetles, weevils, earwigs and ants) (Marchant & Higgins 1993; Jessop 2003; del Hoyo et al. 1996; BirdLife International 2015).

The greater sand plover usually forages visually, with a running, stopping and pecking action typical of many species of plovers. It gleans the surface of the substrate or probes just below the surface (Marchant & Higgins 1993; Jessop 2003).

Migration patterns

After the end of breeding, migratory flocks of the greater sand plover form between mid-June and early-August, and arrive at non-breeding grounds between mid-July and November with adults arriving before juveniles (del Hoyo et al. 1996; BirdLife International 2015). The greater sand plover is often seen migrating in large flocks with lesser sand plovers (Draffan et al. 1983).

The greater sand plover is one of the first migratory shorebirds to return to north-western Australia, usually arriving in late July (Minton et al. 2005a). It is thought that greater sand plovers may make the trip between the breeding grounds and the non-breeding grounds (a distance of ~7,500 km) with only one major stopover (Minton et al. 2006).

The birds who spend the non-breeding period in south-east Asia start moving northwards to the breeding grounds in late-February (the migration peaking in March to early-April), arriving from mid-March to May. Most non-adult birds remain in the southern non-breeding areas during the breeding season (del Hoyo et al. 1996; BirdLife International 2015).

Departure from breeding grounds

The migratory route of the greater sand plover is more westerly than other shorebirds that visit Australia (Minton et al. 2004; Minton et al. 2006). Most band recoveries and flag sighting records have been concentrated in a fairly narrow band in Vietnam, in the southern half of the Chinese mainland, and in Taiwan (Minton et al. 2006). On migration, the species has been recorded only in small numbers in eastern Asia, including eastern and south-eastern China (including Hong Kong), Taiwan and Vietnam (Minton 2005; Ma et al. 2006; Minton 2006; Zheng et al. 2006). However, greater numbers are recorded on passage through south-east Asia, e.g. the Philippines, the Malay Peninsula and Indonesia (Crossland et al. 2006; Bamford et al. 2008).

It has been suggested that greater sand plovers may be capable of non-stop flight between breeding and non-breeding grounds (Marchant & Higgins 1993), which could explain the scarcity of large numbers of greater sand plovers (and "important sites") in east-Asia (Bamford et al. 2008). It may be that sites in south-east Asia, where large numbers have been recorded during southward migration, are the arrival points for birds migrating southwards from the breeding grounds (Bamford et al. 2008). An assessment of the body fat proportions in both adult and juvenile birds considered that greater sand plovers have the ability to fly directly from Taiwan to Australia (Chiang & Liu 2005).

Non-breeding season

The greater sand plover is gregarious during the non-breeding season when it occurs in flocks, sometimes comprising up to several hundred birds (e.g. a single flock of this species at Fog Bay, south-west of Darwin was estimated as 1,800 individuals; Chatto 2005). The greater sand plover often flocks with other shorebirds, especially the lesser sand plover, though the two species usually remain segregated when roosting with one another (Marchant & Higgins 1993).

In Australasia, most records of greater sand plovers during the non-breeding season are from the north coast of Australia, with smaller numbers occurring along other Australian coasts, as well as in Papua New Guinea and New Zealand (Marchant & Higgins 1993). The paucity of inland records within Australia suggests that movements to southern and eastern areas occur around the coastline rather than across the continent, and small numbers migrate through Torres Strait and south along the east coast between September and November (Draffan et al. 1983; Barter & Barter 1988; Marchant & Higgins 1993). The species begins to depart from southern coasts by March, moving north along the east coast, with influxes recorded in Queensland in late March. Birds migrate north through the Top End between late February and April with most adult birds having left the north-west by mid to late April (Barter & Barter 1988; Marchant & Higgins 1993).

Return to breeding grounds

It is considered that a substantial proportion of greater sand plovers departing from Australia have sufficient weight which may enable them to overfly south-eastern Asia and reach the coast of south-west China (Barter & Barter 1988).

Using geolocators, the northward migration of greater sand plovers was tracked from north-west Australia (Broome). The tracked birds appeared to complete large initial flights before stopping in Vietnam or locations further east and then continuing onwards to breeding grounds. All geolocators in this study ceased to function when birds were over north China or Mongolia (Minton et al. 2011). Only a small proportion of greater sand plovers are known to visit the Yellow Sea area. Further geolocator deployments on greater sand plovers will provide more extensive data on stopover locations (Minton et al. 2011).

Threats

Migratory shorebirds, such as the greater sand plover, are sensitive to certain development activities due to their: high site fidelity, tendency to aggregate, very high energy demands required for migration; and need for habitat networks containing both roosting and foraging sites (Department of the Environment 2015a,b).

Threats to the global population of the greater sand plover across its range, but particularly at East Asian staging sites, include: habitat loss and habitat degradation (e.g. through land reclamation, industrial use and urban expansion; reduced river flows; environmental pollution; invasive plants), pollution/contamination impacts, disturbance, direct mortality (e.g. hunting), diseases; and, climate change impacts (Melville 1997; Garnett et al. 2011; BirdLife International 2015; Department of the Environment 2015a,b).

Threats to the greater sand plover in Australia, especially eastern and southern Australia, include ongoing human disturbance, habitat loss and degradation from pollution, changes to the water regime and invasive plants (Garnett et al. 2011; Department of the Environment 2015a,b).

Habitat loss and habitat degradation

There are a number of threats that affect migratory shorebirds in the EAAF with the greatest threat being indirect and direct habitat loss (Melville 1997). As most migratory shorebirds have specialised feeding techniques, they are particularly susceptible to slight changes in prey sources and foraging environments. Activities that cause habitat degradation include (but are not restricted to): loss of marine or estuarine vegetation, which is likely to alter the dynamic equilibrium of sediment banks and mudflats, invasion of intertidal mudflats by weeds such as cordgrass, water pollution and changes to the water regime, changes to the hydrological regime and exposure of acid sulphate soils, hence changing the chemical balance at the site (Department of the Environment 2015a,b).

Migratory shorebird staging areas used during migration through eastern Asia are being lost and degraded by activities which are reclaiming intertidal mudflats for development or converting them for the aquaculture industry (Moores et al. 2008; MacKinnon et al. 2012; Murray et al. 2014).

It is thought that only a small proportion of the EAAF population of greater sand plovers visit the Yellow Sea (Minton et al. 2011). Therefore, compared to a range of other migratory shorebird species that occur in Australia, the greater sand plover may be less likely to have been affected by major loss of intertidal habitat and foreshore reclamation that has been occurring, and continues to occur, in the Yellow Sea region (Minton et al. 2011).

However, habitat loss and intertidal reclamation is also a threat in other areas of the EAAF, such as in Malaysia, where significant numbers of greater sand plovers have been recorded (Wei et al. 2006). In coastal and intertidal areas of Malaysia, migration shorebird habitat is being destroyed or degraded due to land reclamation development activities (e.g. for industries, housing, aquaculture, agriculture and tourism purposes), fishing, logging/destruction of mangroves, and pollution (e.g. domestic sewage, industrial waste, aquaculture waste; Wei et al. 2006).

One of the species' migratory staging areas in China (Chongming Island) is undergoing significant habitat loss and degradation through conversion to aquaculture ponds, farmlands and vegetable gardens, the cultivation of the invasive plant *Spartina alterniflora* on tidal flats (promoting rapid sedimentation with the intention of reclaiming the area), and the Three Gorges Dam on the upper reaches of the Yangtze River reducing the supply of river-borne sediment to mudflats in the area (Ma et al. 2002b; BirdLife International 2015). More than half of all Chinese coastal wetlands were lost between 1950 and 2000 (An et al. 2007). In addition, intensive oil exploration and extraction, and reduction in river flows due to upstream water diversion, are other potentially significant threats in parts of China where this species is present in internationally significant numbers (Barter et al. 1998; Barter 2005).

In Australia, there are a number of threats common to most migratory shorebirds, including the greater sand plover. The loss of important habitat reduces the availability of foraging and roosting sites. This affects the ability of the birds to build up the energy stores required for successful migration and breeding. Some sites are important all year round for juveniles who may stay in Australia throughout the breeding season until they reach maturity. A variety of activities may cause habitat loss at Australian sites. These include direct losses through land clearing, inundation, infilling or draining. Indirect loss may occur due to changes in water quality, hydrology or structural changes near roosting sites (Department of the Environment 2015a,b).

Residential, farming, industrial and aquaculture/fishing activities represent the major cause of habitat loss or modification in Australia (Department of the Environment 2015a,b).The non-breeding grounds of the species in south-eastern Australia are threatened by habitat degradation, loss and human disturbance (Garnett et al. 2011) whereas sites in the Northern Territory are thought to be generally free of such disturbances (Ward 2012).

Climate change

Global warming and associated changes in sea level are likely to have a long-term impact on the breeding, staging and non-breeding grounds of migratory shorebirds (Harding et al. 2007). Migratory shorebirds are also particularly susceptible to heat stress (Battley et al. 2003; Rogers et al. 2006). Climate change projections for Australia include the likelihood of increased temperatures and rising sea levels with more frequent and/or intense extreme climate events which may result in species loss and habitat degradation (Chambers et al. 2005).

Any sea level rise will greatly alter coastal ecosystems, causing habitat change and loss for shorebird species. Modelling has shown that migratory species in the EAAF are at greater risk from sea level rise than previously thought (Iwamura et al. 2013). The modelling indicated that the effect of sea level rise inundating 23–40% of intertidal habitat areas along the migration routes of migratory shorebirds would cause a reduction in population flow (i.e. maximum flow capacity of the migratory population) of up to 72% across the shorebird species assessed. This magnification of effect was particularly due to shorebirds using a few key sites in the EAAF where a large proportion of the population stops and stages (Iwamura et al. 2013).

Pollution/contamination impacts

Migratory shorebirds are also adversely affected by pollution, both on passage and in non-breeding areas (Melville 1997; Harding et al. 2007). Pollution is a particular threat as pollutants tend to accumulate and concentrate in wetlands (Department of the Environment 2015a,b). Industrial pollution (e.g. via accidental release) can lead to the build-up of heavy metals or toxic elements in the substrate of wetlands which, in turn, can affect the benthic prey fauna of shorebirds like the greater sand plover (Department of the Environment 2015a,b).

Disturbance

Human disturbance can cause shorebirds to interrupt their feeding or roosting and may influence the area of otherwise suitable feeding or roosting habitat that is actually used. Disturbance from human recreation activities may force migratory shorebirds to increase the time devoted to vigilance and anti-predator behaviour and/or may compel the birds to move to alternative, less favourable feeding areas (Goss-Custard et al. 2006; Glover et al., 2011; Weston et al., 2012).

Disturbance can result from recreational activities including fishing, boating, four wheel driving, walking dogs, noise and night lighting. While some disturbances may have a low impact, it is important to consider the combined effect of disturbances with other threats (Department of the Environment 2015a,b).

With increasing tourist visitation and development around Broome, Western Australia, increasing levels of disturbance from human recreational activity are likely for the migratory shorebirds in this area. Recreational fishing, four-wheel driving, unleashed dogs and jet-skiing may disturb the foraging or roosting behaviour of migratory shorebirds. Migratory shorebirds are most susceptible to disturbance during daytime roosting and foraging periods (Department of the Environment 2015a,b).

Introduced species

Introduced plants, such as cord grass *Spartinia*, can invade intertidal mudflats and reduce the amount of suitable foraging areas, as has already occurred in other countries (Goss-Custard & Moser 1988). Exotic marine pests may also result in the loss of benthic food sources (Department of the Environment 2015a,b).

Direct mortality

Direct mortality may result from collision with large structures (e.g. wind farms) which cause a barrier to migration or movement pathways, bird strike with vehicles and aircraft, hunting, chemical spills, oil spills and predation (attack by domestic pets, hunting by humans; Schacher et al., 2013; Department of the Environment 2015a,b).

The greater sand plover is subject to commercial hunting (for sale at market or to restaurants) which is a major threat in the area of Chongming Island, China (Ma et al. 2002a; BirdLife International 2015). Records between 1985 and 2009 indicate that at least 567 individuals of this species were hunted in China, Thailand, and Myanmar. Within this period, taking into account the year with lowest take (lower bound) and the year with highest take (upper bound), the possible range of annual take is at least 1 to 340 individuals (Ruttanadakul and Ardseungnerm 1986, Tang and Wang 1995, Ming et al. 1998, Ge et al 2006, Zöckler et al. 2010).

Disease

Since, 1992, the viral disease testing of Charadriiformes from coastal northwest Australia has not detected any evidence of avian influenza virus excretion in the greater sand plover or any other shorebird species tested. However, from serologic testing, there was evidence of a very low level of past exposure to the virus (Curran et al. 2014).

<u>How judged by the Committee in relation to the EPBC Act Criteria and Regulations</u>

Criterion 1. Population size reduction (reduction in total numbers)
Population reduction (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

Evidence:

Eligible under Criterion 1 A2 (a) for listing as Vulnerable

The global population of the greater sand plover has been estimated to be c.180,000 - 360,000 individuals (Wetlands International 2006; BirdLife International 2015). The global population trend for the species is unknown although it is not thought to be decreasing sufficiently rapidly to warrant up-listing from its current global status of 'Least Concern' (BirdLife International 2015). However, the global population trend is difficult to determine because of uncertainty surrounding the impacts of habitat modification on population sizes (BirdLife International 2015).

Of the total global population of 180,000 - 360,000 individuals for the species (Birdlife International 2015), about 125,000–200,000 are thought comprise the subspecies *C. l. leschenaultii*, >10,000 the subspecies *C. l. columbinus*, and about 65,000 the subspecies *C. l. scythicus* (Wiersma 1996).

It has been estimated that ~46% of the global population of the great sand plover occurs in the EAAF (MacKinnon et al. 2012) with about three quarters of the EAAF population occurring in Australia (Bamford et al. 2008). The number of greater sand plovers (all belonging to the subspecies *C. l. leschenaultii*) that occur in the EAAF has been estimated at around 100,000 with approximately 75,000 of these spending the non-breeding period at sites in Australia (Bamford et al. 2008; Garnett et al. 2011).

Numbers of greater sand plovers declined at Moreton Bay, Queensland by c.60% between 1998 and 2008 (Fuller et al. 2009) which has been assessed as a statistically significant decrease of 6% per year (Wilson et al. 2011). Numbers decreased at Eighty-mile Beach, Western Australia by c.65% between 2000 and 2008, whereas numbers at Bush Point were variable between 2004 and 2008 (Rogers et al. 2009; MacKinnon et al. 2012).

Population trends outside Australia are poorly known but numbers in Japan have, in general, slightly increased between 1978 and 2008 (Amano et al. 2010). Overall, the evidence suggests there has been a decline of 30-49% over 17 years across Australia (averaging some contradictory trends) (Garnett et al. 2011). This decline is likely to continue given ongoing threats to this species' migratory staging sites in East Asia (Garnett et al. 2011).

The Committee considers that the species has undergone a very severe reduction in numbers over three generation lengths (24 years for this assessment), equivalent to at least 30-49 percent and the reduction has not ceased, the cause has not ceased and is not understood. Therefore, the species has been demonstrated to have 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 subpopulations; (v) number of mature individuals			
(c) Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations; (iv) number of mature individuals			

Evidence:

Not eligible

The extent of occurrence in Australia is estimated to be 35 700 km^2 (stable) and area occupied 2 600 km^2 (stable; Garnett et al. 2011). Therefore, the species does not meet this required element of this criterion.

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 subpopulation	**≤ 50**	**≤ 250**	**≤ 1,000**
(a)	(ii) % of mature individuals in one subpopulation =	**90 – 100%**	**95 – 100%**	**100%**
(b)	Extreme fluctuations in the number of mature individuals			

Evidence:

Not eligible

The number of mature individuals in Australia was estimated at 75 000 (decreasing) in 2011 (Garnett et al. 2011), but has declined since. Therefore, the species does not meet this required element of this criterion.

Criterion 4. Number of mature individuals			
	Critically Endangered Extremely low	**Endangered Very Low**	**Vulnerable Low**
Number of mature individuals	**< 50**	**< 250**	**< 1,000**

Evidence:

Not eligible

The number of mature individuals in Australia was estimated at 75 000 in 2011 (Garnett et al., 2011), but has declined since. The estimate is not considered extremely low, very low or low. Therefore, the species does not meet this required element of this criterion.

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**

Evidence:

Not eligible

Population viability analysis has not been undertaken

Conservation Actions

Recovery Plan

There should not be a recovery plan for this species, as approved conservation advice provides sufficient direction to implement priority actions and mitigate against key threats. Significant management and research is being undertaken at international, national, state and local levels.

Conservation and Management Actions

- Work with governments along the East Asian – Australasian Flyway to prevent destruction of key breeding and migratory staging sites.
- Protect important habitat in Australia.
- Support initiatives to improve habitat management at key sites.
- Maintain and improve protection of roosting and feeding sites in Australia.
- Advocate for the creation and restoration of foraging and roosting sites.
- Incorporate requirements for greater sand plover into coastal planning and management.
- Manage important sites to identify, control and reduce the spread of invasive species.
- Manage disturbance at important sites which are subject to anthropogenic disturbance when greater sand plovers are present – e.g. discourage or prohibit vehicle access, horse riding and dogs on beaches, implement temporary site closures.

Survey and monitoring priorities

- Enhance existing migratory shorebird population monitoring programmes, particularly to improve coverage across northern Australia.
- Monitor the progress of recovery, including the effectiveness of management actions and the need to adapt them if necessary.

Information and research priorities

- Undertake work to more precisely assess greater sand plover life history, population size, distribution and ecological requirements particularly across northern Australia.
- Improve knowledge about dependence of greater sand plover on key migratory staging sites, and non-breeding sites to the in south-east Asia.
- Improve knowledge about threatening processes including the impacts of disturbance and hunting.

Recommendations

(i) The Committee recommends that the list referred to in section 178 of the EPBC Act be amended by **including** in the list in the Vulnerable category:

Charadrius leschenaultii

(ii) The Committee recommends that there not be a recovery plan for this species.

Threatened Species Scientific Committee

01/03/2016

References cited in the advice

Amano T., T. Székely, K. Koyama, H. Amano & W.J. Sutherland. (2010). A framework for monitoring the status of populations: an example from wader populations in the East Asian–Australasian flyway. *Biological Conservation* 143, 2238–2247.

An, S.Q., H.B. Li, B.H. Guan, C.F. Zhou, Z.S. Wang, Z.F. Deng, Y.B. Zhi, Y.H. Liu, C. Xu, S.B. Fang, J.H. Jiang & H.L. Li. (2007). China's natural wetlands: Past problems, current status, and future challenges. *AMBIO* 36, 335 – 342.

Bamford M., D. Watkins, W. Bancroft, G. Tischler & J. Wahl (2008). *Migratory Shorebirds of the East Asian - Australasian Flyway: Population estimates and internationally important sites*. [Online]. Canberra, ACT: Department of the Environment, Water, Heritage and the Arts, Wetlands International-Oceania. Available from: http://www.environment.gov.au/biodiversity/migratory/publications/shorebirds-east-asia.html.

Barrett, G., A. Silcocks, S. Barry, R. Cunningham & R. Poulter (2003). *The New Atlas of Australian Birds*. Melbourne, Victoria: Birds Australia.

Barter, L., & M. Barter (1988). Biometrics, moult and migration of Large Sand Plover, *Charadrius leschenaulti*, spending the non-breeding season in north-western Australia. *Stilt* 12, 33-40.

Barter, M.A. (1993). Population monitoring of waders in Australia: why is it so important, how is it best done and what can we do?. *Stilt* 22, 13-15.

Barter, M.A., D. Tonkinson, J.Z. Lu, S.Y. Zhu, Y. Kong, T.H. Wang, Z.W. Li & X.M. Meng (1998). Shorebird numbers in the Huang He (Yellow River) Delta during the 1997 northward migration. *Stilt* 33,15-26.

Barter, M.A., K. Gosbell, L. Cao & Q. Xu (2005). Northward shorebird migration surveys in 2005 at four new Yellow Sea sites in Jiangsu and Liaoning Provinces. *Stilt* 48, 13-17.

Battley, P.F., D.I. Rogers, T. Piersa & A. Koolhass. (2003). Behavioural evidence for heat-load problems in Great Knots in tropical Australia fuelling for long-distance flight. *Emu* 103, 97-103.

BirdLife International (2015) Species factsheet: *Charadrius leschenaultii*. Downloaded from http://www.birdlife.org on 06/08/2015.

Carlos, C.J., C.S. Roselaar & J-F. Voisin. (2012). A replacement name for *Charadrius leschenaultia crassirostris* (Severtzov, 1873), a subspecies of Greater Sand Plover. *Bulletin of the British Ornithologists' Club* 132, 63.

Chambers, L.E., L. Hughes & M.A. Weston. (2005). Climate change and its impact on Australia's avifauna. *Emu* 105, 1-20.

Chatto, R. (2003). The Distribution and Status of Shorebirds around the Coast and Coastal Wetlands of the Northern Territory. *Northern Territory Parks and Wildlife Commission Technical Report 73*.

Chatto, R. (2005). A Tour of Selected Significant Coastal Shorebird Sites in the Top End of the Northern Territory of Australia. **In:** Straw, P., ed. Status and Conservation of Shorebirds in the East Asian-Australasian Flyway. Proceedings of the Australasian Shorebirds Conference 13-15 December 2003, Canberra, Australia. Sydney, NSW: Wetlands International Global Series 18, International Wader Studies 17.

Chiang, C.-Y. & W.-T. Liu (2005). Shorebird studies in Taiwan. **In:** Straw,P, ed. Status and Conservation of Shorebirds in the East Asian-Australasian Flyway. Proceedings of the Australasian Shorebirds Conference 13-15 December 2003, Canberra, Australia. Canberra: Australia.

Christidis, L. & W.E. Boles (2008). *Systematics and Taxonomy of Australian Birds*. Collingwood, Victoria: CSIRO Publishing.

Collins, P., A. Boyle, C. Minton & R. Jessop (2001). The importance of inland claypans for waders in Roebuck Bay, Broome, NW Australia. *Stilt* 38, 4--8.

Cramp, S. & K.E.L. Simmons, eds. (1983). Handbook of the Birds of Europe, the Middle East and North Africa. The Birds of the Western Palearctic. Volume 3, Waders to Gulls. Oxford: Oxford University Press.

Crossland, A.C., S.A. Sinambela, A.S. Sitorus & A.W. Sitorus (2006). An overview of the status and abundance of migratory waders in Sumatra, Indonesia. *Stilt* 50, 90-95.

Curran, J.M., T.M. Ellis & I.D. Robertson. (2014). Surveillance of Charadriiformes in Northern Australia shows species variations in exposure to Avian Influenza Virus and suggests negligible virus prevalence. *Avian Diseases* 58, 199-204.

del Hoyo, J., A. Elliott, D.A. Christie & J. Sargatal (1996). *Handbook of the Birds of the World: Hoatzin to Auks*. Barcelona: Lynx Edicions.

Dening, J. (2005). Roost management in south-East Queensland: building partnerships to replace lost habitat. **In:** Straw, P., ed. *Status and Conservation of Shorebirds in the East Asian-Australasian Flyway. Proceedings of the Australasian Shorebirds Conference 13-15 December 2003*. Page(s) 94-96. Sydney, NSW. Wetlands International Global Series 18, International Wader Studies 17.

Department of the Environment (2015a) Wildlife Conservation Plan for Migratory Shorebirds. http://www.environment.gov.au/biodiversity/publications/wildlife-conservation-plan-migratory-shorebirds-2016 (Accessed 07/02/2016).

Department of the Environment (2015b) EPBC Act Policy Statement 3.21 – Industry Guidelines for avoiding, assessing and mitigating impacts on EPBC Act listed migratory shorebird species http://www.environment.gov.au/epbc/publications/shorebirds-guidelines (Accessed 07/02/2016).

Draffan, R.D.W., S.T. Garnett & G.J. Malone (1983). Birds of the Torres Strait: an annotated list and biogeographic analysis. *Emu* 83, 207-234.

Fuller, R.A., H.B. Wilson, B.E. Kendall & H.P. Possingham. (2009). 'Monitoring shorebirds using counts by the Queensland Wader Study Group'. Report to the Queensland Wader Study Group and the Department of Environment and Resource Management, Brisbane.

Garnett, S., J. Szabo & G. Dutson (2011). *The Action Plan for Australian Birds 2010*. CSIRO Publishing.

Ge, Z. M., Wang, T. H., Yuan, X., Zhou, X. and W. Y. Shi. (2006). Use of wetlands at the mouth of the Yangtze River by shorebirds during spring and fall migration. *Journal of Field Ornithology* 77: 347-356.

Gill, F & D Donsker (Eds). (2015). IOC World Bird List (v 5.2). doi : 10.14344/IOC.ML.5.2. http://www.worldbirdnames.org/ (Accessed 06/08/2015).

Gosper, D. & G. Holmes (2002). Status of birds in the Richmond River district, New South Wales, 1973-2000. *Corella* 26, 89-105.

Gosbell, K. & R. Clemens (2006). Population monitoring in Australia: some insights after 25 years and future directions. *Stilt* 50, 162-175.

Goss-Custard, J.D. & M.E. Moser (1988). Rates of change in the numbers of Dunlin, *Calidris alpina*, wintering in British estuaries in relation to the spread of *Spartina anglica*. *Journal of Applied Ecology* 25, 95-109.

Goss-Custard, J.D., P. Triple., F. Sueur & A.D. West. (2006). Critical thresholds of disturbance by people and raptors in foraging wading birds. *Biological Conservation* 127, 88-97.

Harding, J., S. Harding & P. Driscoll (1999). Empire Point Roost: a purpose built roost site for waders. *Stilt* 34, 46-50.

Harding, S.B., J.R. Wilson & D.W. Geering (2007). Threats to shorebirds and conservation actions. **In:** Geering, A., L. Agnew & S. Harding, eds. *Shorebirds of Australia*. Page(s) 197-213. Melbourne, Victoria: CSIRO Publishing.

Huang, S-C., S-S. Shih, Y-S. Ho, C-P. Chen & H-L. Hsieh. (2012). Restoration of Shorebird-Roosting Mudflats by Partial Removal of Estuarine Mangroves in Northern Taiwan. *Restoration Ecology* 20, 76–84.

Iwamura, T., H.P. Possingham, I. Chades, C. Minton, N.J. Murray, D.I. Rogers, E.A. Treml & R.A. Fuller (2013). Migratory connectivity magnifies the consequences of habitat loss from sea-level rise for shorebird populations. *Proceedings of the Royal Society B: Biological Sciences*.

Jessop, A. (2003). Gut analysis of five wader species collected from the nw of Western Australia. *Stilt* 43, 3-13.

Jessop, R. & P. Collins (2000). The effects of cyclonic weather conditions on the bird life around Broome, Western Australia. *Stilt* 36, 11-15.

Ma, Z., C. Choi, X. Gan, S. Zheng & J. Chen (2006). The importance of Jiuduansha Wetlands for shorebirds during northward migration: energy-replenishing sites or temporary stages? *Stilt* 50, 54-57.

Ma, Z.J., K. Jing, S.M. Tang & J.K. Chen. (2002a). Shorebirds in the eastern intertidal areas of Chongming Island during the 2001 northward migration. *Stilt* 41, 6-10.

Ma, Z.J., Tang, S.M., Lu, F. and Chen, J.K. (2002b). Chongming Island: a less important shorebird stopover site during southward migration? *Stilt* 41, 35-37.

MacKinnon, J., Y.I. Verkuil & N. Murray. (2012). IUCN situation analysis on East and Southeast Asian intertidal habitats, with particular reference to the Yellow Sea (including the Bohai Sea). Occasional Paper of the IUCN Species Survival Commission No. 47. IUCN, Gland, Switzerland and Cambridge, UK. ii + 70 pp.

Marchant, S. & P.J. Higgins, eds. (1993). *Handbook of Australian, New Zealand and Antarctic Birds. Volume 2 - Raptors to Lapwings*. Melbourne, Victoria: Oxford University Press.

Melville, D.S. (1997). Threats to waders along the East Asian-Australasian Flyway. **In:** Straw, P., ed. *Shorebird conservation in the Asia-Pacific region*. Page(s) 15-34. Melbourne, Victoria: Birds Australia.

Ming, M., Lu, J. J., Tang, C. J., Sun, P. Y. and W. Hu. (1998). The contribution of shorebirds to the catches of hunters in the Shanghai area, China, during 1997-1998. *Stilt* 33: 32-36.

Minton, C. (2005). What have we learned from banding and flagging waders in Australia? In: Status and Conservation of Shorebirds in the East Asian- Australasian Flyway. P. Straw (ed.). Wetlands International Global Series 18, International Wader Studies 17. Sydney, Australia. pp. 116–142.

Minton, C. (2006). The history of wader studies in north-west Australia. *Stilt* 50, 224-234.

Minton, C., J. Wahl, R. Jessop, C. Hassell, P. Collins & H. Gibbs (2006). Migration routes of waders which spend the non-breeding season in Australia. *Stilt* 50, 135-157.

Minton , C., K. Gosbell, P. Johns, M. Christie, M. Klaassen, C. Hassell, A. Boyle, R. Jessop & J. Fox. (2011). Geolocator studies on Ruddy Turnstones *Arenaria interpres* and Greater Sandplovers *Charadrius leschenaultia* in the East Asian-Australasia Flyway reveal widely different migration strategies . *Wader Study Group Bull*. 118, 87 - 96.

Minton, C., R. Jessop, P. Collins, C. Hassell & L. Beasley (2004). Sightings of waders and terns leg-flagged in north-western Australia: report number 8. *Stilt* 45, 60-70.

Minton, C., R. Jessop, P. Collins, C. Hassell, A. Ewing & H. Gibbs (2005a). Sightings of waders and terns leg-flagged in north-west Australia: report number 9. *Stilt* 47, 47-57.

Minton, C., R. Jessop, P. Collins & C. Hassell. (2005b). NWA 2005 Wader and Tern expedition, 12 February to 6 March 2005. *Stilt* 47, 58-64.

Moores, N., D.I. Rogers, R.-H. Kim, C. Hassell, K. Gosbell, S.-A. Kim & M.-N. Park (2008). *The 2006-2008 Saemangeum Shorebird Monitoring Program Report*. Birds Korea, Busan.

Murray, N.J., R.S. Clemens, S.R. Phinn, H.P. Possingham & R.A. Fuller (2014). Tracking the rapid loss of tidal wetlands in the Yellow Sea. *Frontiers in Ecology and the Environment*. doi:10.1890/130260.

Rogers, D., T. Piersma & C.Hassell. (2006). Roost availability may constrain shorebird distribution: Exploring the energetic costs of roosting and disturbance around a tropical bay. *Biological Conservation* 133, 225-235.

Rogers, D., C.Hassell, J. Oldland, R. Clemens, A. Boyle & K. Rogers (2009). *Monitoring Yellow Sea migrants in Australia (MYSMA): north-western Australian shorebird surveys and workshops, December 2008*.

Ruttanadakul, N. and S. Ardseungnerm. (1986). Evaluation of shorebird hunting in villages around Pattani Bay, Pattani, Thailand. Pp. 152-159. In: Parish, D. and C. Prentice (eds.). Wetland and waterfowl conservation in Asia. Asian Wetland Bureau Publication No. 52. Malaysia.

Straw, P. (1999). Habitat remediation - a last resort?. *Stilt* 35, 66.

Tang, S. X. and T. H. Wang. (1995). Waterbird hunting in East China. Asian Wetland Bureau Publication No. 114. Kuala Lumpur, Malaysia.

Ward, S. (2012). Threatened species of the Northern Territory: Greater Sand Plover *Charadrius leschenaultia*. Northern Territory Department of Land Resource Management. http://www.lrm.nt.gov.au/__data/assets/pdf_file/0015/143124/Greater_Sand_Plover_VU_FINAL.pdf (Accessed 06/08/2015).

Watkins, D. (1993). A national plan for shorebird conservation in Australia. *RAOU Report Series*. 90.

Wei, D.L.Z., Y.C. Aik, L.K. Chye, K. Kumar, L.A. Tiah, Y. Chong & C.W. Mun (2006). Shorebird survey of the Malaysian coast November 2004-April 2005. *Stilt* 49, 7-18.

Wiersma, P. (1996). Charadriidae (Plovers) species accounts. **In:** del Hoyo, J., A. Elliott & J. Sargatal, eds. *Handbook of the Birds of the World. Volume 3. Hoatzin to Auks*. Page(s) 411-442. Barcelona: Lynx Edicions.

Wilson, H.B., B.E. Kendall, R.A. Fuller, D.A. Milton H.P. & Posingham. (2011). Analyzing variability and the rate of decline of migratory shorebirds in Moreton Bay, Australia. *Conservation Biology* 25, 758-766.

Zheng, S., C. Choi, X. Gan, Z. Ma, S. Tang & J. Zhu (2006). Shorebird numbers at the Jiuduansha wetlands during the 2005 southward migration. *Stilt* 50, 58-61.

Zöckler, C., Hla, T. H, Clark, N., Syroechkovskiy, E., Yakushev, N., Daengphayon, S. and R. Robinson. (2010). Hunting in Myanmar is probably the main cause of the decline of the Spoon-billed Sandpiper *Calidris pygmeus*. *Wader Study Group Bull* 117: 1-8.

The Minister approved this conservation advice and included this species in the Vulnerable category, effective from 09/07/2020

Conservation Advice

Falco hypoleucos

Grey Falcon

Taxonomy

Conventionally accepted as *Falco hypoleucos* Gould, 1841. No infraspecific taxa described. The species consists of a single population and is considered monotypic (Marchant and Higgins 1993).

Summary of assessment

Conservation status

Vulnerable: Criterion 4

The highest category for which *Falco hypoleucos* is eligible to be listed is Vulnerable.

Falco hypoleucos has been found to be eligible for listing under the following categories:

Criterion 1: Not eligible
Criterion 2: Not eligible
Criterion 3: Not eligible
Criterion 4: Vulnerable
Criterion 5: Not eligible

Species can be listed as threatened under state and territory legislation. For information on the listing status of this species under relevant state or territory legislation, see http://www.environment.gov.au/cgi-bin/sprat/public/sprat.pl

Reason for conservation assessment by the Threatened Species Scientific Committee

This advice follows assessment of new information provided to the Committee to list Grey Falcon.

Public consultation

Notice of the proposed amendment and a consultation document was made available for public comment for 33 business days between 3 July and 16 August 2019. Any comments received that were relevant to the survival of the species were considered by the Committee as part of the assessment process.

Species/sub-species information

Description

The Grey Falcon is an elusive species endemic to mainland Australia. It is the rarest of six Australian members of the genus *Falco* (Olsen and Olsen 1986; Marchant and Higgins 1993). The Grey Falcon is a medium-sized raptor (400 – 500g) that exhibits reversed sexual dimorphism in body mass, with females weighing on average about 30 per cent more than males (Schoenjahn 2011). The Grey Falcon is a compact, pale grey falcon with a heavy thick chest, long wings and dark wing tips (Debus 2019; Schoenjahn 2010). The under-body is pale grey and the tail has narrow blackish bars. The chin, throat and cheeks are white in colour; adults are pale grey with fine blackish streaks, and juveniles are white with heavy dark streaks.

The legs and toes, eye-ring, cere and base of the bill are bright orange-yellow and the tip of the bill is black (Marchant and Higgins 1993).

Distribution

The species occurs in arid and semi-arid Australia, including the Murray-Darling Basin, Eyre Basin, central Australia and Western Australia (Marchant and Higgins 1993). The species is mainly found where annual rainfall is less than 500 mm, except when wet years are followed by drought, when the species might become marginally more widespread, although it is essentially confined to the arid and semi-arid zones at all times (Schoenjahn 2018).

The species appears to be absent from Cape York Peninsula, areas east of the Great Dividing Range in Queensland and New South Wales, south of the Great Dividing Range in Victoria, and south of latitude 26ºS in Western Australia (Barrett et al. 2003; Schoenjahn 2018).

Relevant biology/ecology

The Grey Falcon occurs at low densities across inland Australia (BirdLife International 2019). The ecology of the Grey Falcon was known almost entirely from anecdotal and opportunistic observations, but has been the subject of significant recent research, especially by Schoenjahn (2011, 2013, 2018) but also by Aumann (2001a,b,c), Falkenberg (2011), Sutton (2011), Watson (2011), Janse et al. (2015) and Ley and Tynan (2016).

The species frequents timbered lowland plains, particularly acacia shrublands that are crossed by tree-lined water courses (Garnett et al. 2011; Watson 2011; Schoenjahn 2013, 2018; Janse *et al.* 2015; Ley and Tynan 2016). The species has been observed hunting in treeless areas and frequents tussock grassland and open woodland, especially in winter (Olsen and Olsen 1986; Schoenjahn 2018).

While breeding Grey Falcons feed almost exclusively on birds (Cupper and Cupper 1980, 1981; Harrison 2000; Aumann 2001c; Falkenberg 2011; Sutton 2011; Schoenjahn 2013; Janse *et al.* 2015; Ley and Tynan 2016). Prey species include doves, pigeons, small parrots and cockatoos, and finches, but a variety of other bird prey species has been recorded (Marchant and Higgins 1993, Hollands 1984; Debus and Rose 2000; Schoenjahn 2013, Cook 2014, Fisher 2015). Non-avian prey recorded by direct observation include small mammals on three occasions (Schoenjahn 2013, Moore 2016) and a lizard (Czechura 1981).

Breeding occurs from June to November. Clutch size can vary from 1 – 4 eggs (Olsen and Olsen 1986; Garnett et al. 2011; Schoenjahn 2013). Eggs are laid in the old nests of other birds, particularly those of other raptors or corvids. The nests chosen are usually in the tallest trees along watercourses, particularly River Red Gum (*Eucalyptus camaldulensis*) and Coolibah (*E. coolabah*), but falcons also nest in telecommunication towers (Marchant and Higgins 1993; Schoenjahn 2013, 2018; Falkenberg 2010). The incubation period is 34–35 days (Cupper and Cupper 1980; Hollands 1984; Sutton 2011; Ley and Tynan 2016) and the nestling period is variously given as 49–52 days (Cupper and Cupper 1980), 41 days (Hollands 1984), 42–49 days (Hollands 2003) and ‘just under 6 weeks’ (Sutton 2011), suggesting that the lower end may be more realistic and in line with other similar-sized Australian falcons. Typically, young Grey Falcons and their parents will stay together for up to at least 12 months after fledging, even when the parents have a new brood (Schoenjahn 2018).

Threats

In the absence of focused studies on Grey Falcons, all potential threats to the species that have been published are based on general considerations and extrapolations from better studied species and are, therefore, speculative (Garnett and Crowley 2000, Garnett et al. 2011). Schoenjahn (2018) identified ten plausible threats to the Grey Falcon and ranked them according to severity (Table 1).

Table 1: Threats impacting the Grey Falcon in approximate order of severity of risk (see Schoenjahn 2018).

Threat factor	Threat status and priority for action	Evidence base
Invasive species		
Predation by cats	Very High	Schoenjahn (2018) documented that Grey Falcons will roost on the bare open ground and documented Grey Falcon in the gut contents of cats. Chicks may be vulnerable to cat predation at accessible nests.
Climate change		
Increased temperatures in arid and semi-arid Australia	Very High	The breeding distribution now covers areas of the highest annual average temperatures in Australia (Schoenjahn 2013). The predicted increases in severity and frequency of days with very high temperatures, heat waves and droughts may exceed the physiological and behavioural capacities of these birds to thermoregulate adequately (Schoenjahn 2018). Changes in rainfall patterns may affect prey availability and heat stress may affect chick survival. However these impacts are speculative and another analysis of climate change impacts on birds did not predict that Grey Falcons would be affected (Garnett et al. 2013; Garnett and Franklin 2014).
Demographic and genetic stochastic events		
Small population size	High	The estimated number of mature individuals is <1,000 (Schoenjahn 2013, 2018; Garnett et al. 2011; BirdLife International 2019). A small population is more susceptible to demographic and genetic stochastic events, which can impact the long term survival of the population.
Habitat loss and fragmentation		
Grazing by exotic herbivores	Very High	Herbivores such as camels in arid and semi-arid areas are preventing the regeneration of suitable nesting trees (Garnett et al. 2011; Schoenjahn 2018). Habitat degradation by herbivores may also reduce prey abundance.
Nest shortage	High	Land clearing of the semi-arid zone and overgrazing of arid zone rangelands have been identified as possible threats to the availability of nesting trees (Garnett and Crowley 2000; Garnett et al. 2011; Schoenjahn 2013, 2018). The loss of artificial structures (telecommunication towers and repeaters) may also contribute to the reduction of suitable nesting habitat (Schoenjahn 2018).
Disturbance		
Birdwatchers and photographers	Moderate	The Grey Falcon is a highly sought after species by birdwatchers and bird photographers. As a consequence, nest sites may be visited by individuals and commercial birding tour groups during the breeding season hoping to see the species. This may cause disturbance and affect breeding success.
Direct mortality		

Collision with traffic	Moderate	Schoenjahn (2018) documented six cases of Grey Falcons being found injured or dead along roads between 2007 and 2017.
Collision with fences and powerlines	Moderate	Grey Falcons have been reported receiving life-threatening injuries from colliding with fences, and presumably power-lines (Schoenjahn 2011).
Harvesting		
Egg collecting	Low	Egg-collecting was considered a threat until the late 1980s (Cupper and Cupper 1981, Dennis 1986, Hollands 1984, SAOA 1992), but may not be of such importance any longer because collecting and possessing eggs without a permit is now illegal in all Australian states and territories.
Falconry	Low	Falconry is illegal in Australia, however, the international demand from falconry for rare falcon species and colour morphs appears to be strong. Schoenjahn (2018) noted that the threat to the Grey Falcon species as a whole from illegal activities in Australia is, at present, minimal.

Threat Prioritisation

Each of the threats outlined above has been assessed to determine the risk posed to the Grey Falcon population using a risk matrix. This in turn determines the priority for actions outlined below. The threats were considered in the context of the current management regimes. The impact of each threat has been assessed assuming that existing management measures continue to be applied appropriately. If management regimes change then the level of risk associated with threats may also change. The risk matrix considers the likelihood of an incident occurring and the consequences of that incident. Threats may act differently in different parts of the species range and at different times of year, but the precautionary principle dictates that the threat category is determined by the population at highest risk. Population-wide threats are generally considered to present a higher risk.

The risk matrix uses a qualitative assessment drawing on peer reviewed literature and expert opinion. In some cases the consequences of activities are unknown. In these cases, the precautionary principle has been applied. Levels of risk and the associated priority for action are defined as follows:

Very High - immediate mitigation action required

High - mitigation action and an adaptive management plan required, the precautionary principle should be applied

Moderate – obtain additional information and develop mitigation action if required

Low – monitor the threat occurrence and reassess threat level if likelihood or consequences change

Table 2: Risk Prioritisation

Likelihood of occurrence	Consequences				
	Not significant	Minor	Moderate	Major	Catastrophic
Almost certain	Low	Moderate	Very High	Very High	Very High
Likely	Low	Moderate	High	Very High	Very High
Possible	Low	Moderate	High	Very High	Very High
Unlikely	Low	Low	Moderate	High	Very High
Rare or Unknown	Low	Low	Moderate	High	Very High

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 basis but only a few times

Rare or Unknown – may occur only in exceptional circumstances; OR it is 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 extinction

Table 3: Grey Falcon Residual Risk Matrix

Likelihood of occurrence	Consequences				
	Not significant	Minor	Moderate	Major	Catastrophic
Almost certain		Birdwatchers and photographers	Predation by cats Increased temperatures in arid and semi-arid Australia Grazing by exotic herbivores		
Likely		Collision with traffic			
Possible		Collision with fences and powerlines	Small population size Nest shortage		
Unlikely					
Rare or Unknown		Egg collecting Falconry			

How judged by the Committee in relation to the EPBC Act criteria and regulations

Criterion 1. Population size reduction (reduction in total numbers) Population reduction (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

Evidence:

Not eligible

No population trend data are currently available. The species occurs at low densities across arid and semi-arid Australia. There is uncertainty about historical declines and recent evidence of declines is lacking (Reid and Fleming 1992; Garnett et al 2011). Garnett et al. (2011) considered that past, present or future population declines are unlikely to exceed 20 per cent in any 3-generation period (18.6 years; Garnett et al. 2011).

Following assessment of the data, the Committee has determined that the species is not eligible for listing in any category under this criterion as the past, current or future population declines are thought unlikely to exceed 30 per cent in any 3-generation period.

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 subpopulations; (v) number of mature individuals			
(c) Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations; (iv) number of mature individuals			

Evidence:

Not eligible

The extent of occurrence (EOO) is estimated at 6.1 million km^2, and the area of occupancy (AOO) estimated at 6,000 km^2 (Garnett et al. 2011). These figures are based on the mapping of point records from post 1997 species observations, obtained from state governments, museums, CSIRO, and Birdlife Australia. 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. Therefore, the species has not met a required element of this criterion.

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 subpopulation	**≤ 50**	**≤ 250**	**≤ 1,000**
	(ii) % of mature individuals in one subpopulation =	**90 – 100%**	**95 – 100%**	**100%**
(b)	Extreme fluctuations in the number of mature individuals			

Evidence:

Not eligible

The species consists of a single population (Marchant and Higgins 1993). The total population size is now generally accepted to be <1,000 mature individuals (Schoenjahn 2011, 2018; Garnett et al. 2011; BirdLife International 2019; Schoenjahn et al. *in press*) and considerably scarcer than previously thought (<5,000 individuals, Brouwer and Garnett 1990; Schoenjahn et al. *in press*). No population trend data are available. There is uncertainty about historical declines and recent evidence of decline is lacking (Reid and Fleming 1992; Garnett et al 2011). Garnett et al. (2011) found no evidence to support a continuing population decline or extreme fluctuations. Therefore, the species has not met a required element of this criterion.

Criterion 4. Number of mature individuals			
	Critically Endangered Extremely low	**Endangered Very Low**	**Vulnerable Low**
Number of mature individuals	**< 50**	**< 250**	**< 1,000**

Evidence:

Eligible under Criterion 4 for listing as Vulnerable

The species occurs at low densities across arid and semi-arid Australia. The species has been encountered very infrequently during extensive, targeted surveys (Schoenjahn 2011, 2018). The total population size is accepted to be <1,000 mature individuals (Schoenjahn 2011, 2018; Garnett et al. 2011; BirdLife International 2019; Schoenjahn et al. *in press*) and considerably scarcer than previously thought (<5,000 individuals, Brouwer and Garnett 1990; Schoenjahn et al. *in press*). This estimate is based on reported range and densities compared with the Peregrine Falcon (*Falco peregrinus*) (reported over two separate time periods 20 years apart for the Atlas of Australian Birds, Blakers *et al.* 1984; Barrett et al. 2003), and assuming 3,000 - 5,000 pairs of Peregrine Falcon in Australia (after Olsen and Olsen 1988).

By comparing the range and number of sightings per 1 degree block in the first Atlas of Australian Birds (Blakers et al. 1984), it is estimated that the Grey Falcon occupies about 0.27× the area occupied by the Peregrine Falcon (99 compared to 365 grid blocks) at an average of one-quarter its density. Given an estimated 3,000–5,000 pairs of Peregrines in Australia (Olsen and Olsen 1988, cited in Garnett et al. 2011), this suggests a total of 200 to 350 pairs of Grey Falcon (Schoenjahn 2011). The second Atlas (Barrett et al. 2003) reports sightings in 118 (14%) compared with 384 (47%) of grid blocks, for the Grey Falcon and Peregrine Falcon respectively. At one-third the distribution and a little over half the density, the estimated population is 550–915 pairs. The average of the mid-point of the ranges from the two Atlases is about 500 pairs and is considered appropriately precautionary, especially considering the uncertainty of the data and historical declines (Garnett et al. 2011), thus the population is estimated at 999 mature individuals. More recent work on the genetic variation of the species is consistent with the <1,000 mature individual estimate (S. Garnett pers. comm. J. Schoenjahn pers. comm.)

The Committee considers that the total number of mature individuals is <1,000 which is low. Therefore, the species has met the relevant elements of Criterion 4 to make it eligible for listing as Vulnerable.

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**

Evidence:
Not eligible

Population viability analysis has not been undertaken.

Conservation actions

Recovery plan

A Recovery Plan is not required; an approved Conservation Advice for the species provides sufficient direction to implement priority actions, mitigate against key threats and enable recovery. Management and research activities are being undertaken at state and local levels.

Primary conservation actions

Support initiatives to improve habitat management, cat and camel control in arid and semi-arid Australia. However, given our understanding of threats is poor, these actions are tentative and may be subject to change in priority.

Conservation and management priorities

- Habitat loss, disturbance and modifications
    - Support improved fire and grazing management in areas where Grey Falcons are known to occur.
    - Protect known nesting trees and include adequate exclusion buffers with regard to proposed developments and land clearing activities.
    - Support the establishment and survival of replacement nest trees in areas where Grey Falcon in known to breed.
    - Retain artificial structures with known or potential Grey Falcon nests.
- Invasive species
    - Control invasive cats and camels in areas where Grey Falcons are known to occur, especially in known roosting and nesting areas.

Stakeholder Engagement

- Engage Indigenous Land Councils, communities, pastoral industry, land managers and non-government organisations to support the conservation of Grey Falcons.
- Discourage the disclosure of locations of active nests to the public.

- Promote the conservation, and raise the profile, of Grey Falcons through strategic programs and educational products with land holders and community groups.
- Promote the exchange of conservation priorities between governments, non-government organisations and communities through use of networks, publications and websites.

Survey and Monitoring priorities

- This species is rare, with a very large distribution. Monitoring population trends is particularly challenging, and will probably require collaboration between many stakeholders to implement, once a suitable approach has been designed.
- Annual surveys of breeding events across the arid and semi-arid zone are recommended including at least the Western Simpson Desert, Tanami Desert and Barkly Tablelands.
- Locating active Grey Falcon nests is aided by:
  - Visiting nests used in previous years;
  - Actively searching for new nests in suitable habitat; and
  - Following up records from the general public, including from Indigenous communities, land managers and bird watchers.

Information and research priorities

- Develop methods for assessing population trends in a rare, widely-distributed species. This requires consideration of logistical, sampling and analytical constraints.
- Continues to collect ecological and demographic information.
- Improve knowledge about potential threatening processes including feral cats, climate change and habitat modification.

Recommendations

(i) The Committee recommends that the list referred to in section 178 of the EPBC Act be amended by **including** in the list in the Vulnerable category:

Falco hypoleucos

(ii) The Committee recommends that there not be a recovery plan for this species.

Threatened Species Scientific Committee

12/09/2019

References cited in the advice

Aumann, T. (2001a) Breeding biology of raptors in riparian environments in the south-west of the Northern Territory, Australia. *Emu* 101: 305–315.

Aumann, T. (2001b) Habitat use, temporal activity patterns and foraging behaviour of raptors in the south-west of the Northern Territory, Australia. *Wildlife Research* 28: 365–378.

Aumann, T. (2001c) An intraspecific and interspecific comparison of raptor diets in the south-west of the Northern Territory, Australia. *Wildlife Research* 28: 379–393.

Barrett, G., Silcocks, A., Barry, S., Cunningham, R. and Poulter, R. (2003). The New Atlas of Australian Birds. Birds Australia (Royal Australasian Ornithologists Union), Melbourne.

BirdLife International (2019) Species factsheet: *Falco hypoleucos*. Downloaded from http://www.birdlife.org on 07/05/2019.

Blakers, M.; Davies, S. J. J. F.; Reilly, P. N. (1984). The atlas of Australian birds. Royal Australasian Ornithologists' Union, Victoria.

Brouwer, J. and Garnett, S. (eds) (1990) Threatened birds of Australia – an annotated list. RAOU Report No. 68 Royal Australasian Ornithologists' Union, Melbourne.

Cook, D. (2014). A tern up for the books. Australian Birdlife 3: 62.

Cupper, J. and Cupper, L. (1980). Nesting of the Grey Falcon *Falco hypoleucos*. Australian Bird Watcher 8: 212–219.

Cupper, J. and Cupper, L. (1981). Hawks in Focus. Jaclin, Mildura Vic., Australia.

Czechura, G. V. (1981). Some notes on the Grey Falcon. Australasian Raptor Association News 2(2): 9–10.

Debus, S. (2019). Birds of Prey of Australia: A Field Guide, 3rd edn. CSIRO Publishing, Melbourne.

Debus S.J.S., and Rose, A.B. (2000) Diet of Grey Falcons *Falco hypoleucos* breeding extralimitally in New South Wales. *Australian Bird Watcher* 18: 280–281.

Dennis, T. (1986). South Australian area co-ordinator's report for 1985. Australasian Raptor Association News 7(2): 24–25.

Falkenberg, I. D. (2011). Aspects of the ecology of the Grey Falcon *Falco hypoleucos* in the South Australian arid zone. Corella 35: 23–28.

Fisher, K. D. (2015). Grey Falcon eating Australian Pratincole. Boobook 33: 51.

Garnett, S. T. and Crowley, G. M. (2000). The Action Plan for Australian Birds 2000. Environment Australia, Canberra.

Garnett, S.T. and Franklin, D.C. (2014) Climate change adaptation plan for Australian birds. CSIRO Publishing. Collingwood, Australia.

Garnett, S.T., Franklin, D.C., Ehmke, G., VanDerWal, J.J., Hodgson, L., Pavey, C., Reside, A.E., Welbergen, J.A., Butchart, S.H.M., Perkins, G.C. and Williams, S.E. (2013) *Climate change adaptation strategies for Australian birds*, National Climate Change Adaptation Research Facility, Gold Coast, 925 pp.

Garnett, S. T., Szabo, J. K. and Dutson, G. (2011). The Action Plan for Australian Birds 2010. CSIRO, Melbourne.

Harrison, R. (2000). Observations on the Grey Falcon *Falco hypoleucos*. Australian Bird Watcher 18: 267–269.

Hollands, D. (1984). Eagles, Hawks and Falcons of Australia. Nelson Australia, Melbourne.

Hollands, D. (2003). Eagles, Hawks and Falcons of Australia, 2nd edn. Bloomings Books, Melbourne.

Janse, I., Kloecker, U., Roshier, D. and Witte, I. (2015) Breeding diet and behaviour of a pair of Grey Falcons Falco hypoleucos and their offspring in north-western New South Wales. Corella 39: 46–51.

Ley, A. and Tynan, B. (2016) Observations on nesting Grey Falcons, *Falco hypoleucos*. South Australian Ornithologist 41: 49–64.

Marchant, S. and Higgins, P. J. (Eds) (1993). Handbook of Australian, New Zealand & Antarctic Birds, Vol. 2: Raptors to Lapwings. Oxford University Press, Melbourne.

Martin, W. K. and Royal, M. J. (2001). Easterly records of the Grey Falcon in New South Wales. Australian Bird Watcher 19: 132–134.

Moore, E. D. (2016). Grey Falcon *Falco hypoleucos* taking a small mammal as prey. Australian Field Ornithology 33: 30–31.

Olsen, P. D. and Olsen, J. (1986). Distribution, status, movements and breeding of the Grey Falcon *Falco hypoleucos*. Emu 86: 47–51.

Reid, J. and Fleming, M. (1992). The conservation status of birds in arid Australia. Rangeland Journal 14: 65–91.

SAOA [South Australian Ornithological Association] (1992). Threatened birds of South Australia. Pp. 6.1–6.11 in Tay, S. P. (Ed.), Threatened species and habitats in South Australia - a catalyst for community action, Conservation Council of South Australia, Adelaide.

Schoenjahn, J. (2010). Field identification of the Grey Falcon *Falco hypoleucos*. Australian Field Ornithology 27: 49–58.

Schoenjahn, J. (2011). Morphometric data of recent specimens and live individuals of the Grey Falcon *Falco hypoleucos*. Corella 35: 16–22.

Schoenjahn, J. (2013). A hot environment and one type of prey: investigating why the Grey Falcon (*Falco hypoleucos*) is Australia's rarest falcon. Emu 113: 19–25.

Schoenjahn, J. (2018). Adaptations of the rare endemic Grey Falcon *Falco hypoleucos* that enable its permanent residence in the arid zone of Australia. PhD Thesis. University of Queensland

Schoenjahn, J., Pavey, C.R. and Walter, G.H. (*in press*) Ecology of the Grey Falcon *Falco hypoleucos* – current and required knowledge. Emu

Sutton, A.J.G. (2011) Aspects of the biology of the Grey Falcon *Falco hypoleucos* in the Pilbara region of Western Australia. Corella 35, 11–15.

Watson, C. (2011) A failed breeding attempt by the Grey Falcon *Falco hypoleucos* near Alice Springs, Northern Territory. Australian Field Ornithology 28, 167–179.

Australian Government

Department of Agriculture, Water and the Environment

Conservation Advice for *Phascolarctos cinereus* (Koala) combined populations of Queensland, New South Wales and the Australian Capital Territory

In effect under the *Environment Protection and Biodiversity Conservation Act 1999* from 12 February 2022.

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

Phascolarctos cinereus (Koala) © Copyright Karen Ford.

Contents

Conservation status

The *Phascolarctos cinereus* (Koala) combined populations of Queensland, New South Wales and the Australian Capital Territory were determined to be a species for the purposes of the *Environment Protection and Biodiversity Conservation Act 1999* (Cwth) (EPBC Act) (s517) on 27 April 2012 and listed in the Vulnerable category of the threatened species list under the EPBC Act effective from 2 May 2012. For conciseness, it is referred to hereafter as the listed population.

The listed population was reassessed in 2021 by the Threatened Species Scientific Committee to be eligible for listing as Endangered under criteria 1. The Committee's assessment is at Attachment A. The Committee's assessment of the listed population's eligibility against each of the listing criteria is:

- Criterion 1: A2C and A4C: Endangered
- Criterion 2: Ineligible
- Criterion 3: Ineligible
- Criterion 4: Ineligible
- Criterion 5: Insufficient data

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

Species information

Taxonomy

This species is conventionally accepted as *Phascolarctos cinereus* (Koala) (Goldfuss 1817). The koala is a single species whose physical appearance differs with latitude. Morphological differences include size, fur colouration and fur length. Three subspecies of koala were previously described on the basis of morphological differences in size and fur colouration: *Phascolarctos cinereus adustus* (Queensland) (Thomas 1923), *P. c. cinereus* (New South Wales) (Goldfuß & Bischof 1817) and *P. c. victor* (Victoria) (Troughton 1935). There is no genetic evidence to support these subspecies (Wedrowicz et al. 2017). A recent genomic assessment of population structure indicates spatially-organised genetic structure within the species (Kjeldsen et al. 2019), meaning that a large proportion of genetic variation can be attributed to the geographic distance between populations (Eldridge & Lott 2020).

The koala population was sub-divided in 2012 due to substantial differences in management and conservation status across the species range. Under section 517 of the EPBC Act the combined koala populations of Queensland, New South Wales and the Australian Capital Territory were declared to be a species for the purposes of the Act. This entity was listed as Vulnerable. The koala was not found to be eligible for listing at the national scale (SEWPaC 2012b).

Description

Names applied to the koala by different Indigenous language groups include: Guula and Gulawayn in the Gathang language, Barrandhang, Gurabaan, Naagun and Ginaagun from the Wiradjuri language, Borobi in the Yuambeh language, Doombearpee and Dumirripi in the Jandai language, and Goala from the Kabi language.

The koala is a medium-sized marsupial with a stocky body, large, rounded ears, sharp claws and variable but predominantly grey-coloured fur. Males are typically larger than females. Its morphological appearance changes gradually from south to north across its range, with larger individuals in the south and smaller individuals in the north. The average weight of males is 12 kg in Victoria compared with 6.5 kg in Queensland. In the south, the koala is characterised by longer, thicker, brown-grey fur, whereas in the north it has shorter, silver-grey fur (Martin & Handasyde 1999).

Distribution

The National distribution

The koala is a wide-ranging marsupial endemic to Australia. It typically occurs in eastern Australian forests and woodlands of predominantly *Eucalyptus* species. Its historical range extends over 22° of latitude and 18° of longitude (Martin & Handasyde 1999). The koala's distribution is not continuous across this range and it occurs in several subpopulations that are separated by cleared land or unsuitable habitat (Martin and Handasyde 1999; NSW DECC 2008). The koala's distribution includes Queensland, New South Wales, the Australian Capital Territory, Victoria and South Australia. The listed population of the koala has a wide but patchy distribution that spans the coastal and inland areas of Queensland north to the Herberton area, extending westwards into hotter and dryer semi-arid climates of central Queensland, New South Wales and the Australian Capital Territory.

Other populations, which are not listed as threatened under the EPBC Act, occur to the south, in Victoria and South Australia. The species is widespread in lowland and foothill eucalypt forests and woodlands across Victoria. Its distribution extends to the south-east corner of South Australia. A number of successful introductions have expanded the distribution in South Australia to locations including Kangaroo Island and mainland areas of the Adelaide Hills, Eyre Peninsular and sites along the Murray River.

The natural range of the koala is determined by specialist food, habitat and environmental requirements. Typically, this includes forests and woodlands dominated by *Eucalyptus* species (Melzer et al. 2000). The koala's home range (the area an individual needs to survive) is highly variable and dependant on life history stage, soil fertility, habitat quality and nutritional requirements. Consequently, home ranges across the species' distribution are highly variable, with home ranges in Queensland and New South Wales reported to vary between 3 and 500 ha (home range data summarised in: Wilmott 2020). Habitat suitability models indicate that koalas are best suited to locations where the mean maximum summer temperatures are 23-26^{o}C and mean annual rainfall ranges from 700 -1500 mm (Adams-Hosking et al. 2011). However, koalas can occur in more extreme environments at the limits of their natural range (McAlpine et al. 2015).

The koala's distribution and population size have declined significantly since European colonisation (Melzer et al. 2000; Sherwin et al. 2000). Much of the koala's national distribution now overlaps with human-modified landscapes. Vegetation clearance from activities including urbanisation, grazing, agriculture and mining have significantly reduced the koala's distribution (McAlpine et al. 2015). Climate change drivers (e.g., drought and rising temperatures) have also resulted in a reduction in climatically suitable habitat (Adams-Hosking et al. 2011).

Concerns over the declining koala population in Queensland, New South Wales and the Australian Capital Territory resulted in the koala combined populations of Queensland, New South Wales and the Australian Capital Territory being listed as Vulnerable in 2012 under the EPBC Act (TSSC 2012). Historically, national and regional estimates of koala population size have been limited, fragmented and based on limited data (Melzer et al. 2000). This has made quantitative assessment of koala populations at the national level problematic. The 2012 listing highlighted the lack of peer-reviewed population data (TSSC 2012). In response to this data gap, in 2012, an expert elicitation estimated that the national koala population size was 329,000 (range: 144,000-605,000) (Adams-Hosking et al. 2016). It also indicated a 24 percent decline nationally over the preceding three koala generations (15-21 years). For the listed population in Queensland and New South Wales, the percentage loss was estimated at 53 percent and 26 percent respectively. No data were detailed for the Australian Capital Territory, however the earlier 2012 listing advice (TSSC 2012) suggested a high likelihood of the koala being present in the Australian Capital Territory, though with some populations originating from deliberate introductions from outside the Australian Capital Territory and possibly some natural populations.

For the listed population of koalas in Queensland, New South Wales and the Australian Capital Territory, extent of occurrence (EOO), the area encompassing all known occurrences of a species across its range (IUCN 2019), is estimated to be 1,665,850 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 (DAWE, 2020). During the 2019-2020 bushfire season an estimated 9 percent (>36,800 km^2) of the koala's distribution was impacted by fire (DAWE 2021a). This agrees with estimates generated by the NESP Threatened Species Recovery Hub of 9-11.4 percent.

In contrast to the Queensland, New South Wales and Australian Capital Territory populations, koala populations in the southern part of the species' range, in Victoria and South Australia, are robust, and in some cases overpopulation has led to active population control measures being put in place (Menkhorst 2008). Despite historically suffering from population crashes and relocations, koala numbers are currently high in Victoria (Heard and Ramsey 2020) and mainland South Australia (DEW 2018), and those subpopulations are not listed.

Koala translocations have occurred in areas outside their natural range. These have resulted in establishment of new populations both in mainland areas (e.g. Adelaide Hills, Eyre Peninsula, Riverland) and on many islands in South Australia (Kangaroo Island), Victoria (French Island, Phillip Island, Raymond Island, Snake Island) and Queensland (Brampton Island, Magnetic Island, St Bees Island) (Melzer et al. 2000). Koalas have also been re-introduced to areas within their natural range in the Australian Capital Territory, New South Wales, mainland Victoria and the south-east of South Australia.

For the listed population in Queensland, New South Wales and the Australian Capital Territory, modelling of koala distribution indicates that in future it will be further constrained by climatic stressors (Adams-Hosking et al. 2011). In particular, shifts in summer temperatures, humidity and water availability pose a significant threat to the koala as a result of acute physiological stress during heatwaves, compounded by drought (Runge et al. 2021a). Forecasting models predict that a large area of koala habitat may be lost, accompanied by a large reduction in the koala population, under 2070 climate change projections (Adams-Hosking et al. 2011; Runge et al. 2021b). These losses will result in the southwards and eastwards contraction of suitable habitat across their range. Models indicate that koala occupancy is strongly dependant on annual rainfall and the distance to water features (Santika et al. 2014). Koalas may survive in refuge areas where microclimates such as deep gullies, caves, cliffs or dense vegetation provide refuge from heat, and perennial water results in leaf-water content remaining high (Runge et al. 2021a).

Distribution across the range of the EPBC Act listed koala population: Queensland, New South Wales and the Australian Capital Territory

Queensland distribution

Koalas are widespread across Queensland (map 1), occurring in patchy and often low-density populations across the different bioregions. They occur as far north as the Einasleigh Uplands and Wet Tropics bioregions with records to the south and west in the Desert Uplands, Central Mackay Coast, Mitchell Grass Downs, Mulga Lands, Brigalow Belt North, Brigalow Belt South, and South Eastern Queensland where they are most frequently sighted (Adams-Hosking et al. 2016). Koalas in Queensland inhabit the moist coastal forests, southern and central western subhumid woodlands, and a number of eucalypt woodlands adjacent to waterbodies in the semi-arid western parts of the state (Melzer et al. 2000). In many locations, koala populations are of low density, widespread and fragmented (Melzer et al. 2018). Surveys in north-western Queensland found that koalas were patchily distributed, associated with creek-lines, areas of higher tree species richness, with higher abundance correlating with leaf-moisture content (Munks et al. 1996).

State-wide estimates of population size are limited, with data and survey effort skewed towards south-east Queensland. In response to this, and the lack of peer reviewed estimates of koala numbers highlighted in the 2012 listing advice (SEWPaC 2012a; TSSC 2012), an expert elicitation exercise was undertaken in 2012 (Adams-Hosking et al. 2016). The data from this expert elicitation are now widely recognised as the most accurate baseline for koala population numbers across the bioregions, states and territories (e.g., NSW Government 2020; Dissanayake et al. 2021) and therefore supersede the 2012 listing data. These data provide a reference point for this Conservation Advice. In 2012, this expert elicitation estimated that that there were 79,264 koalas in Queensland distributed across 8 bioregional areas (Adams-Hosking et al. 2016). The highest population estimates were reported for three bioregions: Brigalow Belt North (15,179), Mulga Lands (15,286) and South East Queensland (15,821). The other bioregions with koalas present included Central Mackay Coast (8857), Desert Uplands (6357), Einasleigh Uplands and Wet Tropics (4750), Mitchell Grass Downs (1943), South Brigalow (11,071). In 2012, it was estimated that Queensland’s koala populations had declined over the three preceding generations (15 to 21 years) by an average of 53 percent (Adams-Hosking et al. 2016).

The eight Queensland bioregions with koalas cover a total area of 1,489,650 km^2. This represents a mean density of 0.0005 koalas/ha across the 8 bioregions in Queensland based on 2012 population estimates. Across the state, South East Queensland has the most comprehensive dataset, reflecting higher survey effort. Based on 2012 population estimates (Adams-Hosking et al. 2016), the bioregions with the highest density of koalas in Queensland included the Central Mackay Coast (0.006 koalas/ha) and South East Queensland (0.002 koalas/ha). Both these bioregions were impacted by bushfire in the 2019-2020 bushfires. In 2021, within the eight Queensland bioregions, an estimated 13 percent (194,021 km^2) of land area overlapped with the koala species distribution model (DAWE 2021a). Of this, 1,931 km^2 of modelled likely koala distribution burnt across the state in the 2019-20 bushfires, representing a total 1 percent of modelled likely koala distribution (DAWE 2021a). Four bioregions were impacted by fire: South East Queensland (2 percent burnt), Central MacKay Coast (2 percent), Brigalow Belt South (1 percent burnt), and New England Tablelands (1 percent). Modelling of future climate-suitable koala distribution indicates a further contraction of 17 to 78 percent by 2030 from the 2011 baseline as a direct result of climate change (Adams-Hosking et al. 2011; Adams-Hosking et al. 2016). The bioregions predicted to be most heavily impacted by climate change included the Mulga Lands (100 percent of climatically suitable koala habitat lost by 2030), the Desert Uplands (100 percent loss by 2030) and the Central Mackay Coast (57 to 96 percent loss by 2030).

New South Wales distribution

Koalas in New South Wales occur from the northern border with Queensland. The northern NSW distribution includes the Mulga Lands, Darling Riverine Plains, Brigalow Belt South, Nandewar, New England Tablelands, and South East Queensland (NSW Section) bioregions. Koalas also occur within the eastern coastline bioregions of the NSW North Coast, Sydney Basin and South East Corner at the border with Victoria. Their western distribution extends into the South-Eastern Highlands, NSW South Western Slopes, Cobar Peneplain, Riverina, and Murray Darling Depression bioregions (Map 1). Koalas occupy a wide range of habitats (NSW Government 2019b, a). The majority of koalas in New South Wales are found in forests and subhumid woodlands on the central and north coast , and to the west across the Western Plains and slopes, within Pilliga forest, low woodland and forested areas (TSSC 2012; Adams-Hosking et al. 2016). Low-density populations also occur west of the Great Dividing Range in semi-arid environments. Habitat in these areas is fragmented and this has resulted in a patchy distribution of koalas across their range with significant numbers occurring on privately owned land (Melzer et al. 2000; Lunney et al. 2009; TSSC 2012). Modelling of koala habitat in New South Wales suggests climate-suitable habitat will contract by 8 to 19 percent by 2030 from the 2011 baseline as a direct result of climate change (Adams-Hosking et al. 2011; Adams-Hosking et al. 2016). Koala distribution has shrunk across NSW, with declines documented from the eastern coastal bioregions to the western populations (Predavec et al. 2018). These declines have been driven by habitat loss, temperature increase and drought (Lunney et al. 2014; Santika et al. 2015). Extinction risk is predicted to be greater in western NSW than in the east under future scenarios of climate and land use change (Santika et al. 2014). Predicted changes in the near (2030) and more distant (2070) future include increased maximum temperatures, reduced minimum temperatures, more extremely hot days (where maximum temperature > 35^{o}C), shifting rainfall patterns, and an increase in average fire weather days. Modelling indicates that by 2070 the habitat losses will be severe (NSW Government 2014).

In 2012, the mean population estimates for koalas within bioregions indicated that the highest numbers of individuals occurred in the bioregions of South Brigalow and Nandewar (11,133), NSW North Coast (8,367) and the Sydney Basin (5,667) (Adams-Hosking et al. 2016). Other bioregions had smaller, but significant koala populations (<3,000 individuals): Murray-Darling Depression (55), South East corner (655), Cobar Peneplain and Riverina (2,354), Darling-Riverine Plains (9,964), Mulga Lands (711), New England Tablelands (2,771), NSW Southwestern Slopes (2,310), South-Eastern Highlands (1363). This study concluded that the NSW koala population had declined by over 26 percent in the preceding (and potentially future) three koala generations (Adams-Hosking et al. 2016).

In 2018, the NSW Framework for the spatial prioritisation of koala conservation actions (Rennison & Fisher 2018) concluded that both the expert elicitation data (Adams-Hosking et al. 2016) and the available records trend data indicated a significant decline in koalas across the state in recent years. The only bioregion to have convincing evidence of a stable population was the New England Tablelands. Since this framework was developed, this bioregion has been impacted by bushfire (see below).

Across the 15 bioregions in NSW containing koalas, nine were impacted by the 2019-20 bushfires with a total of 34,666 km^2 burnt (DAWE 2021). The bioregions most heavily impacted by fire included the South East Corner (52 percent burnt), the Sydney Basin (30 percent burnt) and NSW North Coast (30 percent burnt). Other bioregions that contain koalas and were significantly burnt are: South Eastern Queensland (NSW section) (19 percent burnt), South Eastern Highlands (13 percent burnt), New England Tablelands (13 percent burnt), Australian Alps (4 percent burnt), Nandewar (4 percent burnt), and NSW South Western Slopes (2 percent burnt). Koalas have displayed nuanced responses to fire with significant declines in numbers following high severity fire but little change in occupancy or density following low severity fire (NSW Government 2021a). Further research is required to understand how fire impacted koalas across the different bioregions.

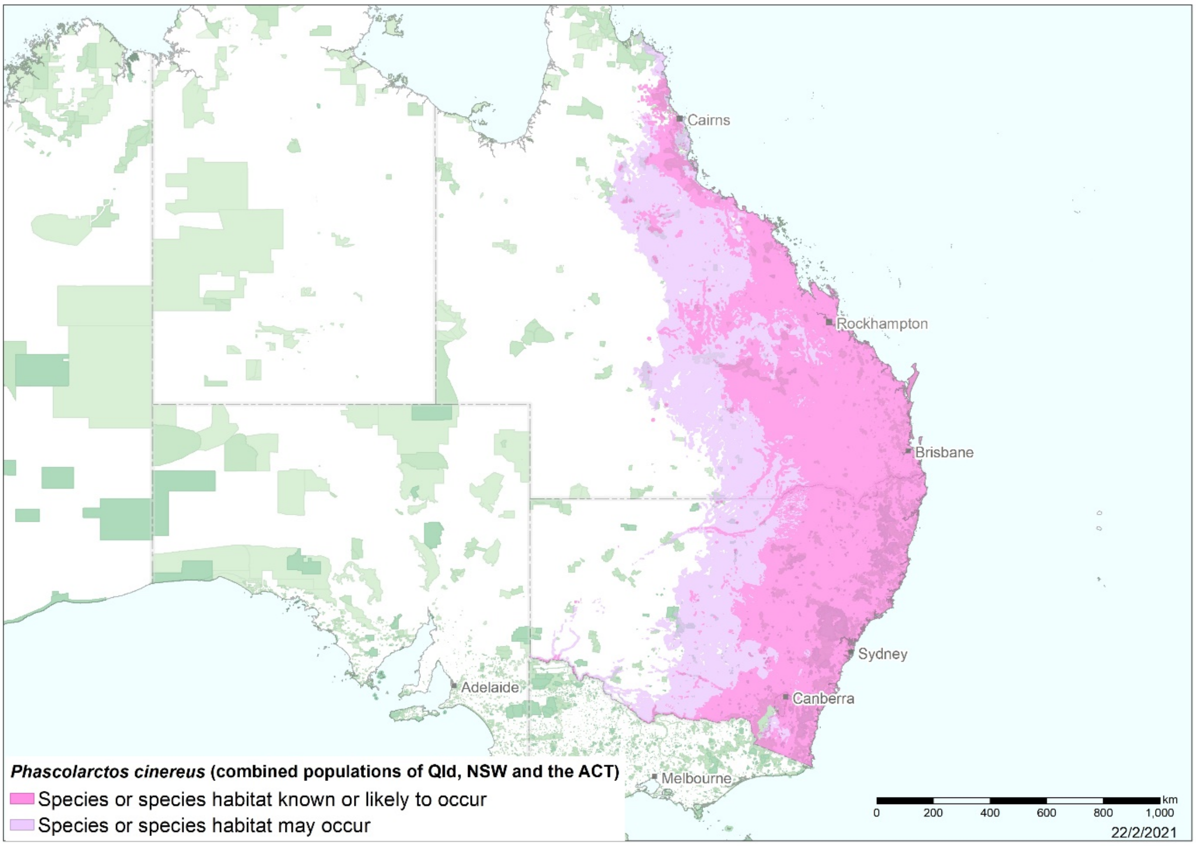
The Australian Capital Territory distribution

Koalas have historically occurred in the Australian Capital Territory. In 2009, it was suggested that small koala populations were historically present in the Tidbinbilla and Brindabella Ranges, around Bushfold, the Orroral Valley and Namadji National Park (TSSC 2012). These populations were thought to be the result of deliberate introductions as well as remnant, natural koala populations. In the 2012 expert elicitation process the Australian Capital Territory was not considered separately and Australian Capital Territory data were aggregated into NSW estimates (Adams-Hosking et al. 2016).

There have been limited reports of koalas in the Australian Capital Territory along the border with New South Wales. In May 2021 a solitary koala was observed over several days in Oaks Estate near the Molonglo River (K Ford, 2021 pers com May 11). In 2014 a koala was observed crossing the highway close to Defence land, near Canberra airport (Fitzgerald 2014). There are also historic records in the 1980s of koalas on the western borders of the Australian Capital Territory and it was suggested that these were animals dispersing from Brindabella. A koala survey in 2018 was conducted in areas considered to be likely koala habitat and no koalas were recorded (Capital-Ecology 2018). The site selection was based on ACT koala survey guidelines. However, thirteen hard-to-access monitoring sites, which included seven sites in Namadgi National Park, plus an additional fifteen Commonwealth owned Defence sites were not included in the survey. The report recommended that acoustic surveys be conducted in the breeding season to confirm these findings (Capital-Ecology 2018). Currently there are no known resident koala populations and koala surveys are not routinely conducted in the Australian Capital Territory.

The bioregions which contain koala habitat in the Australian Capital Territory include large areas that have been impacted by bushfire. In particular, the Orroral Valley, a location where koalas have historically been observed, burnt in 2003 and again in 2019-2020. In the 2019-20 bushfires, an estimated 23 percent (211 km^2) of koala habitat burned (DAWE 2021a). Koala habitat occurs in two bioregions in the Australian Capital Territory, the Australian Alps and South Eastern Highlands, of which 57 percent (102 km^2) and 15 percent (109 km^2) respectively of the total area burnt in recent bushfires. Modelling suggests climatically suitable koala habitat in the Australian Capital Territory will contract by 10 percent by 2030 from the 2011 baseline as a direct result of climate change (Adams-Hosking et al. 2011; Adams-Hosking et al. 2016).

Map 1 Modelled species distribution of the listed koala in Queensland, New South Wales and the Australian Capital Territory. Note that the listed koala distribution does not include Victoria or South Australia.



Source: Draft base map Geoscience Australia; species distribution data Species of National Environmental Significance database. The 2021 SDM was modelled using Maxent, with the harmonised habitat mapping subsequently incorporated (Runge et al. 2021b).

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 the recent observed locations of the species (known to occur), b) the suitable or preferred habitat occurring in close proximity to these locations (likely to occur); and c) 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 observation records, national and regional-scale environmental data, environmental modelling techniques and documented scientific research.

Cultural and community significance

Koalas are culturally significant for many Indigenous peoples across south-eastern and eastern Australia. They hold a significant and diverse role in many Indigenous cultural practices and belief systems. The koala's name has many interpretations within the different Indigenous languages. The word koala may be a loan word derived from *gula* or *gulawan* from the Dharuk language of the Sydney region (Cahir et al. 2020). Early western spellings also include "coola" and koolah". The name "Koala" may also reflect the fact that koalas rarely or never drink free water. Other local Indigenous names such as "*kaola*" translate as "no drink". Several Indigenous narratives describe the koala as the giver or taker of water (Cahir et al. 2020).

The cultural and community significance of the koala is specific and unique to different Indigenous language groups. In New South Wales, koalas are prominent in creation stories and narratives and are known to be totemic for different language groups (Cahir et al. 2020). They are depicted in rock art and were hunted for meat prior to the arrival of colonists. The skins of the koala were used to make rugs by the Gumbaynggirr peoples, while Elders in the Goulburn Plains region used koala fur in initiation ceremonies (Cahir et al. 2020). In Queensland, their spiritual significance can be linked to epic creation stories while in certain regions koalas were hunted for their skin and fur (Cahir et al. 2021). In Victoria, koalas also have a utilitarian and symbolic significance being a revered animal. Records from the region suggest that in some areas they were traditionally used for food but not for skins or fur (Schlagloth et al. 2018). The historic relationship between Aboriginal communities and koalas across the listed range highlights the importance of consulting with Aboriginal communities when planning and undertaking koala focused conservation activities (Cahir et al. 2021).

The koala forms an integral part of modern Australian identity. From the first colonial exhibits in 1861 at the Melbourne Zoo to today, it has become a national icon that is recognised internationally as a symbol of Australia (Markwell 2020a, b). The koala is depicted widely in art, children's books, television shows and popular culture. Many celebrities opt to be photographed with koalas and they have been used widely in marketing campaigns (e.g., Qantas Airlines from 1967 to 1992) (Markwell 2020b). While considered by many Australians as an intrinsically valuable component of Australian fauna, the koala also contributes significantly to tourism.

Relevant biology and ecology

Female koalas reach sexual maturity between 2 and 3 years of age (McLean & Handasyde 2007) and may then produce one offspring per year. Females have a 12-month lactation period and young koalas are weaned after this period. Weaning coincides with periods of high food availability and favourable climatic conditions. This ensures the best survival conditions for offspring approaching independence (Ballantyne et al. 2015). Local factors, including population density, food quality and availability, soil type and climate, influence the timing of breeding (McLean & Handasyde 2007; Ballantyne et al. 2015). Koalas may not breed every year if conditions are unfavourable, and breeding can be unsuccessful due to poor body condition or disease (e.g. *Chlamydia*) (McLean & Handasyde 2007).

Koala reproduction is heavily influenced by seasonality, and the breeding season differs between northern and southern populations. In the north, an estimated 60 percent of births occur in summer and early autumn (December-March), and the remainder are distributed throughout the year (Ellis et al. 2010a). The trigger for this increase in birth rate is not known but does coincide with periods of peak rainfall in Queensland. It has been suggested that opportunistic breeding occurs when food availability increases as a response to rainfall (Ellis et al. 2010a). In locations where rainfall is less seasonally variable, joeys are produced at any time of the year. In South Australia, the ratio of male to female births has also been shown to vary with half of male births occurring before the end of November. In contrast, 50 percent of female births do not occur until the end of December (McLean & Handasyde 2007). One explanation for this is that females in good condition, with greater resource availability, produce larger, healthier male offspring due to an increased period of maternal investment. Studies report no evidence of sex ratio differences in the timing of births, or the size of joeys in Queensland (Ellis et al. 2010a).

In the wild, longevity is more than 15 years for females and more than 12 years for males (Martin & Handasyde 1999). Generation length is defined here as: "*the average age of parents of the current cohort (i.e., newborn individuals in the population)*" (IUCN 2019). The generation length of the listed koala is therefore estimated to be 6 to 8 years. This is also consistent with other assessments (Phillips 2000; TSSC 2012; Woinarski 2020).

Koalas are tree-dwelling, obligate folivores (leaf eaters) with a highly specialised diet. The koala's diet is defined by the availability and palatability of a limited variety of *Eucalyptus*, *Corymbia* and *Angophora* species. Koalas are nocturnal and spend significant periods of time moving across the ground between food and shelter trees. Movement increases in the breeding season (typically September to February) (Melzer & Tucker 2011). Koalas are reported to utilise more than 400 different species of tree for their food and habitat requirements with different tree species varying by habitat type and location across their range. Primary food species differ across habitats and may be as few as two at a particular location (Melzer et al. 2000; Tucker et al. 2008; Kjeldsen et al. 2019). Koala browsing preferences show regional differences which are influenced by the chemical profiles and water content of different target food leaves. There is both intra- and inter-species variability in the palatability and nutritional value of the leaves of their preferred food trees (Stalenberg et al. 2014). Their specialist dietary requirements determine their potential habitat and range distributions (Moore & Foley 2005; Moore et al. 2010).

Habitat critical to the survival

Koala habitat includes both coastal and inland areas that are typically characterised by Eucalyptus forests and woodlands. The wide-ranging distribution of the koala has resulted in a diverse range of habitat associations across different bioregions, influenced by local climate, topographical and landscape associations. Biophysical habitat attributes for the koala include places that contain the resources necessary for individual foraging, survival (including predator avoidance), growth, reproduction and movement. The total amount of resources (including habitat attributes) and how they are arranged in the landscape influence the viability of metapopulations and processes.

For an individual koala, these resources include access to sufficient quality food and shelter trees to meet their daily energetic requirements and reproductive needs, and a place to avoid predators. This includes forests or woodlands, road-side and rail vegetation and paddock trees, safe intervening ground matrix for travelling between trees and patches to forage and shelter and reproduce and access to vegetated corridors or paddock trees to facilitate movement between patches. These resources fall within individual koala's home ranges and allow for interaction with adjacent individuals.

A population of koalas requires a sufficient total amount of resources within their habitat of adequate quality to support a viable biological population where mortality, survival, and recruitment are balanced or recruitment increasing to optimal carrying capacity and within the bounds of natural fluctuations. Crucial habitat elements include patches and corridors for gene flow. Over longer-time frames habitat critical includes climate refugia such as drainage lines, riparian zones and patches that are resilient to drying conditions due to favourable hydrological systems. Additionally, it includes areas that may be temporarily unoccupied, because of seral (maturity or time) changes to habitat quality that arise through processes such as fire, drought, timber harvesting or disease (shifting habitat mosaic) or degradation and are available for future recolonisation.

Habitat critical to the survival of a species is defined as: the areas that the species relies on to avoid or halt decline and promote the recovery of the species. Under the EPBC Act, the following factors and any other relevant factors may be considered when identifying habitat that is critical to the survival of a species:

(a) whether the habitat is used during periods of stress (examples: flood, drought or fire);

(b) whether the habitat is used to meet essential life cycle requirements (examples: foraging, breeding, nesting, roosting, social behaviour patterns or seed dispersal processes);

(c) the extent to which the habitat is used by important populations;

(d) whether the habitat is necessary to maintain genetic diversity and long-term evolutionary development;

(e) whether the habitat is necessary for use as corridors to allow the species to move freely between sites used to meet essential life cycle requirements;

(f) whether the habitat is necessary to ensure the long-term future of the species or ecological community through reintroduction or re-colonisation;

(g) any other way in which habitat may be critical to the survival of a listed threatened species or a listed threatened ecological community.

Such areas, if identified, would be expected to include habitat occupied and habitat currently unoccupied, areas necessary for population processes and maintenance of genetic diversity and evolutionary potential, and areas required to accommodate future population increase, recolonisation, reintroduction, or as climate refugia.

The information set out in this conservation advice relating to the functional ecology of the koala and its habitat are likely to form the basis of habitat critical to the survival of the koala. Having regard to the above factors and other relevant factors at the time of completing this document, it is not practicable to identify by description and to provide spatial information on the habitat critical to the survival of the koala. This is because there is insufficient knowledge and data to unambiguously identify and spatially delineate habitat critical to the survival of the koala. A National Koala Monitoring Program was established in 2021 in response to these critical data requirements.

This document provides general guidance for habitat critical to the survival of the listed koala. The EPBC Act referral guidelines are available for potential proponents to navigate the complexity of koala habitat to identify significant impacts (DofE 2014). The guidelines provide guidance on important requirements, survey planning, and standards for mitigation impacts in context of long-term recovery planning for the listed koala.

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.

Important populations are defined as those that are valued for cultural, social, and economic reasons as well as for the species conservation.

i) For conservation of the listed koala, among other reasons, it will be imperative to maintain populations that:

    - have the potential to act as source populations to adjacent areas of suitable, or potentially suitable, habitat;
    - exist in areas of climatically suitable refugia during periods of environmental stress including droughts, heatwaves, and long-term climate change;
    - are genetically diverse;
    - are disease free and/or exhibit low rates of infection with important pathogens;
    - contain genes which may confer adaptation to current and future environmental stressors;
    - are geographical or environmental outliers within the species range.

ii) Populations are also valued for social, cultural or economic reasons, and may or may not, overlap with populations listed above. Reasons may include, but not limited to:

    - cultural and spiritual importance to Indigenous people;
    - the social value and enjoyment of having koalas close to residential areas;
    - the economic value brought to local business and tourism;
    - the iconic species value at the national and international political and community level.

State level important populations

At the state and territory level, New South Wales has identified critical koala populations as “areas of currently known high koala occupancy” (DPIE 2020). Queensland has identified priority areas for management actions to achieve the highest likelihood of conservation outcomes for koalas in South East Queensland. This has included prioritising koalas located in high quality habitat with a high likelihood of successful threat management (DES 2020). Current efforts to assess and identify important populations across the range are hindered by a lack of comprehensive, unbiased data (DPIE 2020; DAWE 2021b) with the majority of study effort focusing on high density koala populations in easily accessible locations. The 2021 National koala Monitoring Program will address these critical data gaps. Examples of important populations are detailed below.

Genetically important populations

Four spatially distinct, genetic koala management units have been identified nationally (Johnson et al. 2018; Eldridge & Lott 2020). These important genetic populations include: 1) Queensland and New South Wales populations north of the Clarence River Valley, New South Wales; 2) south of the Clarence River Valley, New South Wales to north of the Sydney Basin; 3) south of the Sydney Basin to approximately the New South Wales /Victorian boarder; and 4) Victoria and South Australia populations. Work on the genetic values of different populations is still in its infancy and research is ongoing.

Climate sensitive populations

Koalas at the western edge of their range are being impacted by shifts in rainfall patterns and increasing frequency of drought and heat stress resulting directly from climate change (Adams-Hosking et al. 2011; Davies et al. 2013; Runge et al. 2021b). The recent national workshop of koala monitoring experts (DAWE 2021b) identified the koala subpopulations at the western edges of Queensland and New South Wales distributions (western edge populations) as a priority for immediate climate-related risk management and conservation efforts. The western edge populations are characterised by low koala densities and a high level of isolation from other populations, as a result of which they are increasingly vulnerable to environmental change and habitat loss. The western-edge populations were identified as potentially containing adaptive genes to environmental stressors indicating they have high conservation value (K Handasyde 2021, pers comm 9 February). The workshop recommendations included: an urgent need for population and ecological data (e.g. fertility rates, longevity, movement patterns, habitat requirements, thermal ecology); research into heat tolerance; action to protect these populations as they may prove critical to New South Wales and Queensland in the future; and consideration of translocation of individuals from these genetically important reservoir population to create an insurance population that could prove critical to future management.

Other important populations

Populations that have the potential to act as source populations for adjacent areas of suitable habitat and/or potentially suitable habitat. This includes climate-robust populations, large populations that exist in contiguous habitats, and populations that may link two larger populations.

Threats

The koala is threatened by wide-scale climate change drivers which include the increased frequency and intensity of drought and high temperatures, the increasing prevalence of weather conditions which promote bushfire, and a shrinking climatically suitable area (Adams-Hosking et al. 2011; McAlpine et al. 2015; Runge et al. 2021a). Simultaneously, koala populations are also being impacted by diseases, specifically koala retrovirus (KoRV) and Chlamydia (*Chlamydia pecorum*), human-related activities including habitat loss resulting from land clearance and mining, and mortality due to encounters with vehicles and dogs. These threats can also act synergistically. For example, habitat clearance and climate change drivers are associated with increased levels of physiological stress in wild koala populations (Narayan 2019). This in turn can increase the incidence and impact of localised threats arising from encounter mortality with dogs and vehicles, disease, and food shortages (Narayan 2019).

Table 1 Threats impacting the koala

Threat	Status and severity [a]	Evidence
Climate change driven processes and drivers		
Loss of climatically suitable habitat	• Status: current and future • Confidence: known • Consequence: severe • Trend: increasing • Extent: across the entire range	Areas that are climatically suitable for koalas are contracting (Adams-Hosking et al. 2011). Climate change predictions indicate drier, warmer conditions across the koala's range. Current and future climate change projections indicate a progressive eastward and southwards contraction in the koala's suitable climate envelope and consequent suitable habitat (Adams-Hosking et al. 2011). Modelled climatic suitability from 2010 to 2030 indicates a 38-52% reduction for the listed population (Adams-Hosking et al. 2011), and forecast a 62% decline in koala habitat by 2070. This represents a 79% loss in Queensland and 31% loss in New South Wales (Runge et al. 2021a). The effects of climate change may play out through increased mortality associated with heat wave events and droughts, declines in reproduction rates associated with changes in food quality and availability, changes to movement patterns, exposure to diseases and other factors, as well as effects of climate change on fire regimes (see below for further details on these mechanisms).
Increased intensity/frequency of drought	• Status: historical, current and future • Confidence: known • Consequence: severe • Trend: increasing • Extent: across part of its range	Over the last 21 years, South East Australia has experienced two of its worst droughts in the historical record: the Millennium Drought (2000-2009) and the Big Dry (2017-2019). Low rainfall has been linked with physiological stress to koalas due to low moisture levels, causing negative effects on population viability (Davies et al. 2013). These periods of abnormally low rainfall, particularly in the west, have been associated with the decline, and in some cases, the crash of koala populations, forcing population contraction to critical riparian areas in some areas (Seabrook et al. 2011; DPIE 2020). In extreme cases, e.g., Springsure in Central Queensland, the areas worst affected by drought were along creeks where extensive

Threat	Status and severity [a]	Evidence
		tree death (die back) occurred and negatively impacted koala populations (Ellis et al. 2010b). In the future, average winter and spring rainfall are predicted to continue to decline across the koala's range (BoM 2021a). By the late twenty-first century, the frequency of moderate, severe, extreme and exceptional terrestrial water storage droughts is projected to increase substantially due to a reduction in the frequency of near-normal and wet conditions in Australia (Pokhrel et al. 2021). Cumulative frequency of droughts across the koala range are projected to increase by 30% by 2100 under RCP6.0 (the climate pathway we are on) (NOAA 2021). The frequency of severe and extreme droughts (Drought Severity Index >-1.6) will increase from 2.7% to 19.5%. This suggests that koala habitat will be in drought half the time, and severe drought every 5 years, on average. This is an increase from the currently observed frequency of drought every 5 years and severe drought every 30 years. Droughts also interact with threats posed by inappropriate fire regimes.
Increased intensity/frequency of heatwaves	• Status: historical/current/future • Confidence: known • Consequence: severe • Trend: increasing • Extent: across the entire range	Heatwaves can be defined as ≥ 3 consecutive days of unusually high night-time and day-time temperature (BoM 2021b). Due to climate change, average temperatures across the koala's range will continue to increase across all seasons resulting in an increased frequency and intensity of heat stress days and heat wave episodes (BoM 2021a). Heat stress threats will synergistically interact with drought, further exacerbating the impacts of reduced water availability. During periods of extreme heat stress koalas are also known to stop eating and starve to death (K Youngentob, pers comm 22/3/21).
Increased intensity/frequency of bushfire	• Status: historical/current/future • Confidence: known • Consequence: severe • Trend: increasing • Extent: across part of its range	During the summer of 2019-2020, > 3.5 million ha of koala habitat burnt across Queensland, New South Wales and the Australian Capital Territory (DAWE 2021a). Recent estimates suggest a population decline of 10% (or as much as 17%, with 80% confidence) one year after the 2019-20 bushfires. Of this, a decline of 7.1% was directly caused by bushfires, the remaining 2.3%

Threat	Status and severity [a]	Evidence
		decline was due to ongoing and antecedent threats (Legge et al. 2021). The future legacy of the 2019-2020 bushfires, assuming no future extreme events over three generations (2021-2042), indicates a population decline of 3.9% caused by the fires; a further population decline of 21.9% is attributed to antecedent and ongoing threats (Legge et al. 2021). Koala monitoring records from north-east New South Wales following the 2019/2020 bushfires, indicate that sites characterised by high-severity fire (e.g., canopy scorch) had zero koala occupancy (i.e., zero return/recovery) immediately post fire. At sites where koalas have been detected following fire, refuge areas were present in the surrounding landscape, or fire severity was lower (NSW Government 2021b). While koala's have returned to bushfire impacted locations it is likely to take many years before populations are fully re-established. Australia will continue to experience a harsher fire-weather climate into the future (BoM 2019, 2021a). The fire season length is increasing and the number of catastrophic fire days will increase in the future by an estimated 15-70% by 2050 (Climate-Council 2019). A broad range of fire-related threats exist. These include high frequency fire, high severity fire, shifts in fire season, biodiversity loss, declining ecological mechanisms, shifts in biotic interactions including reproduction and fire-predator interactions, fire-drought interactions, fire-fragmentation interactions which can be amplified by land clearing and logging, fire-climate feedback (see above) (Bradshaw et al. 2018; Leavesley et al. 2020). All of these threats will have a significant impact on koala habitat and resident populations.

Threat	Status and severity [a]	Evidence
Declining nutritional value of foliage	• Status: historical/current/future • Confidence: suspected/known • Consequence: severe • Trend: increasing • Extent: across part of its range	In-situ carbon dioxide (CO_2) manipulation experiments on *Eucalyptus tereticornis* and *E. amplifolia* found elevated CO_2 levels caused total nitrogen to decline in young eucalyptus leaves (Wujeska-Klause et al. 2019). However, increases in environmental temperature (eT), that will occur in parallel with elevated CO_2 in the future, were not included in open air experiments and green house experiments suggest eT may compensate completely for the negative impacts of CO_2 on leaf nitrogen in the future (DeGabriel et al. 2009; Robinson et al. 2012). Although elevated CO_2 can influence the production of some plant secondary metabolites such as tannins that may also impact the digestibility of leaves, the evidence for this in eucalypts is equivocal and further research is needed. Additional research is required to assess how elevated levels of CO_2 affect nitrogen and available nitrogen (which integrates the effects of tannins) (DeGabriel et al. 2009). Bushfire effects on the nutritional value of eucalypt regrowth (e.g., epicormic growth) are unknown and research has been initiated. Physical disturbance (e.g., logging during forestry activities and/or fire) alters tree species composition and can favour tree species that do not support the koala's nutritional requirements (Au et al. 2019).
Human related activities		
Clearing and degradation of koala habitat	• Status: historical, current and future • Confidence: known • Consequence: severe • Trend: increasing • Extent: across the entire range	Human activities (e.g., deforestation and land clearance for grazing, agriculture, urbanisation, timber harvesting, mining and other activities) have resulted in habitat loss, fragmentation and degradation. Over 10,000 km^2 of forest and woodland within the koala's range was cleared between 2000 and 2017 (Ward et al. 2019). The modelled koala distribution was revised in 2021 and the estimate of habitat loss would be expected to be higher if calculated using the new understanding of koala distribution. Clearing for grazing during this period was the major driver of loss of koala habitat, accounting for most of

Threat	Status and severity [a]	Evidence
		the deforestation within koala distribution (McAlpine et al. 2015; Evans 2016). Clearing for grazing has occurred across the range of the koala. Large areas of woodland have been lost since 2000 in western parts of the species range, including the Brigalow Belt, Mulga Lands, Darling Riverine Plains, Einasleigh Uplands and Desert uplands (Ward et al. 2019). These bioregions are home to large koala populations (Adams-Hosking et al. 2016). Most clearing has occurred on freehold or leasehold land (Ward et al. 2019). Land clearing continues to impact habitat across the koala's range (DES 2018). Clearing for mining and urbanisation has had localised impacts on the koala (Evans 2016; Ward et al. 2019). Urban expansion is concentrated along the eastern seaboard fringe of Queensland and NSW (Clark & Johnston 2016), which is also a stronghold of the koala. Low density and peri-urban development are expanding into forested and agricultural landscapes in these areas, while clearing for agriculture continues to occur across the koala's distribution. The expanding coal and coal seam gas developments of the past two decades and recent clearing for renewable energy projects represent additional but localised impacts to koalas (McAlpine et al. 2015). Land-use decisions affecting koalas have been influenced, both positively and negatively, by the policy environment and social attitudes around land-clearing (Heagney et al. 2021; Simmons et al. 2021).
Encounter mortality with vehicles and dogs	• Status: historical/current/future • Confidence: known • Consequence: severe • Trend: increasing • Extent: across part of its range	Vehicle related mortality occurs regularly on roads in close proximity to occupied koala habitat (Gonzalez-Astudillo 2018; Queensland-Government 2021). Dog attacks are also a significant cause of death and injury especially in areas within and adjacent to peri-urban and residential areas (DPIE 2020). Koalas are unable to adapt to these threats and as human activities continue to expand into koala habitat, trauma from these threats will continue. A large proportion of individuals killed by vehicles or dogs are otherwise healthy. This mortality has the potential to remove healthy

Threat	Status and severity [a]	Evidence
		breeding individuals from the population (Gonzalez-Astudillo 2018). Encounter mortality poses a significant threat during post-weaning dispersal, which occurs at a young age in both male and female koalas. Mature males are increasingly at risk as they have larger home ranges and increased mobility during the breeding season. Young males typically disperse more frequently and over larger distances than their female counterparts and the removal of subadult males by trauma has the potential to critically disrupt geneflow.
Disease and health		
Koala retrovirus (KoRV) and Chlamydia (*Chlamydia percorum*)	• Status: historical/current/future • Confidence: known • Consequence: severe • Trend: increasing • Extent: unknown	Wild populations carry disease pathogens. Inadvertent spread of disease also occurred historically following koala translocations. Disease can be a major contributor to population decline and reduces population viability. Chlamydia causes infertility, blindness and death (Polkinghorne et al. 2013). The prevalence of disease (chlamydiosis) has been found to increase following extreme stress from hot weather, drought, habitat loss and fragmentation (Lunney et al. 2012; Davies et al. 2013).

Status—identify the temporal nature of the threat;

Confidence—identify the extent to which we have confidence about the impact of the threat on the species;

Consequence—identify the severity of the threat;

Trend—identify the extent to which it will continue to operate on the species;

Extent—identify its spatial content in terms of the range of the species.

Each threat has been described in Table 1 in terms of the extent that it is operating on the species. 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 the experts listed in DAWE (2021b) and in-house expertise using available literature.

Table 2 Risk matrix

Likelihood	Consequences				
	Not significant	**Minor**	**Moderate**	**Major**	**Catastrophic**
Almost certain	Low risk	Very high risk **Encounter mortality with**		Very high risk **Clearing of koala habitat**	Very high risk **Shrinking climate envelope**

Likelihood	Consequences				
	Not significant	Minor	Moderate	Major	Catastrophic
		vehicles and dogs		**koala retrovirus (KoRV) and Chlamydia** **Increased frequency of drought** **Increased frequency of heatwaves** **Increasing frequency of high-intensity bushfire**	**resulting in habitat loss**
Likely	Low risk	Moderate risk	High risk	Very high risk	Very high risk
Possible	Low risk	Moderate risk	High risk	Very high risk	Very high risk
Unlikely	Low risk	Low risk	Moderate risk	High risk	Very high risk
Unknown	Low risk	Low risk	Moderate risk	High risk	Very high risk

Threats in the above matrix have been classified according to likelihood of threat across the entire range of the listed koala population. It should be noted that at a smaller scale (e.g., regional scale) the risk of individual threats may be classified elsewhere in the table.

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 ties

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’.

Conservation and recovery actions

The following conservation and recovery actions are identified and should be adhered to in conjunction with the latest guidance documents available on the Species Profile and Threats Database:

http://www.environment.gov.au/cgi-bin/sprat/public/publicspecies.pl?taxon_id=85104.

Four supporting strategies and two on-ground (direct) strategies are included.

Supporting strategies:

Strategy 1: **Build and share knowledge**

Strategy 2: **Strong community engagement and partnerships**

Strategy 3: **Increase habitat protection**

Strategy 4: **Koala conservation is integrated into policy, and statutory and land-use plans**

On-ground strategies:

Strategy 5: **Strategic habitat restoration**

Strategy 6: **Active metapopulation management**

Supporting strategies provide for governance to coordinate actions, led by the Australian Government in partnership with the states and territory. They provide for research and capacity building to improve effectiveness of actions, from enhanced mapping, monitoring and survey methods; improved data collation, curation and analysis; to better sharing and communication of information; and building on community capacity, support and engagement. They also provide for improved planning frameworks and principles for state-level conservation planning for the listed koala.

Increasing the area of priority koala habitat that is protected is a key strategy to prevent further habitat loss and fragmentation and prevent further loss of koala populations. Once identified (Actions 1a-c), national areas of priority koala habitat should include areas of large intact landscapes that have the greatest potential to retain viable populations and have the potential to also act as source populations to adjacent areas.

On-ground (direct) strategies relate to improving habitat quality and restoration, and the suite of collective actions required to ensure metapopulation processes are maintained. The former will generally be implemented at the site-level, while the latter is a holistic landscape-scale approach to metapopulation management.

Many state-level actions have been ongoing, or recently commenced, under various state and territory environment-related, or koala-specific strategies (DES 2020; DPIE 2020).

Priorities assigned to actions under each of the six strategies are interpreted as follows:

Priority 1: Urgent. Prompt action is needed in advance of implementation of other management actions, to ensure effective coordination or to provide crucial information for planning and management. Early action might also be necessary to avoid or mitigate the most significant threats.

Priority 2: Essential. Action is necessary to avoid or mitigate direct threats, implement planning and management, undertake research, and develop tools towards the long-term recovery.

Priority 3: Highly beneficial. Action is desirable, and while not critical, will provide for longer term maintenance of recovery.

Strategy 1: Build and share knowledge

The actions here comprise knowledge-based inputs or activities that support direct conservation actions. These inputs will provide information for a strategic and coordinated approach to koala conservation, now and into the near future using predictive climate change impacts. Without actioning these inputs, the ability to implement an effective listed koala recovery, will be significantly diminished.

Action No.	Description	Potential Partners/Responsibility	Timeframe (Priority)	Est. cost
1 a	Identify nationally important populations and habitat across the listed koala range under current conditions, and considering future impacts of climate change such as drought, heatwave, and fire, assessed by undertaking habitat distribution, population modelling and analysis (including abundance/density and genetic diversity), allowing for iterative updates using a robust scenario-based approach	Coordinated by the Commonwealth with state and territory government agencies using internal OR external mapping and modelling experts OR Expert Technical Advisory Panel of the National koala Recovery Team to be formed when the recovery plan is made OR researchers	Year 1 (1)	To be assessed (TBA)
1 b	Identify spatially and temporally strategic areas of high priority for: (i) restoration and revegetation based on koala and eucalypt population viability; (ii) climate and fire refugia; and (iii) corridors facilitating movement and metapopulation processes of koalas, allowing for iterative updates using a robust scenario-based approach.	Coordinated by the Commonwealth with state and territory government agencies, local government and natural resource management organisation Or NGOs Or researchers.	Year 1 and ongoing (1)	TBA
1 c	Develop prioritisation at regional or other appropriate scales for the long-term implementation of actions.	Coordinated by the Commonwealth with state and territory government agencies using internal OR external mapping and modelling experts OR Expert Technical Advisory Panel of the National koala Recovery Team to be formed when the recovery plan is made OR researchers	Year 2 (2)	TBA

Action No.	Description	Potential Partners/Responsibility	Timeframe (Priority)	Est. cost
1 d	In consultation with each range state and territory, including Victoria and South Australia, scope out and establish a fit-for-purpose long-term National koala Monitoring Program (NKMP) to improve understanding of trends in populations, distribution and population health across the koala's range, and efficacy of management interventions.	Coordinated by the Commonwealth with state and territory government agencies; community groups; non-government conservation organisations; koala research community; koala welfare organisations and the Expert Technical Advisory Panel of the National koala Recovery Team to be formed when the recovery plan is made	Year 1-2 (1)	$ 2 million
1 e	Implement National koala Monitoring Program; review design to ensure it remains fit-for-purpose and adaptive	Coordinated by the Commonwealth with state and territory government agencies; community groups; non-government conservation organisations; koala research community; koala welfare organisations and the Expert Technical Advisory Panel of the National koala Recovery Team to be formed when the recovery plan is made	Year 1 and ongoing (1)	TBA
1 f	Collate and synthesise existing data that may improve understanding of koala population dynamics and threat profiles across habitats and scales.	Coordinated by the Commonwealth with state and territory government agencies using internal OR external mapping and modelling experts OR Expert Technical Advisory Panel of the National koala Recovery Team to be formed when the recovery plan is made OR researchers	Years 1-5 (1)	TBA
1 g	Mapping of key metrics (distribution, habitat restoration, habitat condition and habitat loss) is reviewed at appropriate timeframes to detect changes, is coordinated across jurisdictions, and provides for landscape management now and at least three koala generations into the future.	Coordinated by the Commonwealth with state and territory government agencies using internal OR external mapping and modelling experts OR Expert Technical Advisory Panel of the National koala Recovery Team to be formed when the recovery plan is made OR researchers	Ongoing (1)	TBA
1 h	Coordinate pre-existing national and koala databases; coordinate and develop data standards (including metadata standards); survey and sampling design standards to improve the quality of koala monitoring (e.g., Community of Practice).	Coordinated by the Commonwealth with state and territory government agencies; koala research community; koala welfare organisations and the Expert Technical Advisory Panel of the National koala Recovery Team to be formed when the recovery plan is made	Years 1-5 (2)	TBA

Action No.	Description	Potential Partners/Responsibility	Timeframe (Priority)	Est. cost
1 i	Establish national research priorities targeted at applied outcomes, that inform and improve koala management. This action builds on priority research identified by Expert Technical Advisory Panel and the outputs of the first koala expert elicitation workshop for NSW (Hemming et al. 2018).	Coordinated by the Commonwealth with state and territory government agencies; koala research community; koala welfare organisations and the Expert Technical Advisory Panel of the National koala Recovery Team to be formed when the recovery plan is made	Ongoing (2)	TBA
1 j	Establish a recurring research forum to enhance existing collaboration among researchers, and between researchers, managers and other interested parties, to make the most effective use of research actions and to identify and address any further key knowledge gaps.	Coordinated by the Commonwealth with state and territory government agencies and Expert Technical Advisory Panel	Annually (2)	TBA
1 k	Facilitate a network to establish and support an active National koala Recovery Team and Expert Technical Advisory Panel, with strong governance in place.	Coordinated by the Commonwealth with state and territory government agencies	Year 1 (1)	TBA
1 l	Share knowledge across experts, government organisations, conservation groups, rescue and welfare groups, Indigenous groups and the general public through regular koala workshops and conferences. This includes a koala conference every five years that brings together researchers, policy makers, planners and interested conservation groups and citizens; and exceptional circumstance workshops, such as following responses after major crises (e.g., fire and drought).	Coordinated by the Commonwealth with state and territory government agencies and Expert Technical Advisory Panel of the National koala Recovery Team to be formed when the recovery plan is made	5 yearly (3)	TBA
1 m	Facilitate the ongoing capture, storage and subsequent sharing, including intergenerational transfer, of Traditional Knowledge on the koala within the Indigenous community and across civil society. Build and demonstrate the strong connection to koalas and their habitat maintained by Indigenous Australians (e.g., https://koala.nsw.gov.au/culture/)	Coordinated by Indigenous Australians in partnership with Commonwealth, State and Territory government agencies, NGOs and philanthropists	Ongoing (1)	TBA

Strategy 2: Strong community engagement and partnerships.

Successful koala conservation relies on a collaborative approach across all sectors, and communities have a key role to play in protecting local koalas. The high level of community support for the conservation of koalas provides an opportunity for a range of actions that contribute to shared goals, from formal partnerships for habitat protection to raising awareness. Actions include engaging citizens in koala conservation science, supporting and training professionals and koala carers in the community.

Action No.	Description	Potential Partners/Responsibility	Timeframe (Priority)	Est. cost
2 a	Grow partnerships with Indigenous and community groups and local government organisations to co-design opportunities for citizens to be involved in long-term koala monitoring programs and research.	Commonwealth, state and territory government resource in coordination with natural resource management organisations; National koala Recovery Team; Indigenous organisations; NGOs and the Zoo and Aquarium Association.	Ongoing (1)	TBA
2 b	Grow partnerships with Indigenous and community groups, non-government organisations and all level of governments to restore priority areas using best-knowledge revegetation guidelines for koala.	Commonwealth, state and territory government agencies in coordination with natural resource management organisations; National koala Recovery Team; Indigenous organisations and NGOs	Ongoing (1)	TBA
2 c	Develop active communication, education and extension strategies for businesses (developers, industries and rural land-owners' enterprises) aimed at koala habitat protection, incentives, partnership and compliance.	Commonwealth, state and territory government agencies in coordination with local government, natural resource management organisations	Ongoing (2)	TBA

Action No.	Description	Potential Partners/Responsibility	Timeframe (Priority)	Est. cost
2 d	Recognise the cultural and spiritual importance of the koala to Indigenous communities and engage to utilise, improve or reinvigorate their support and knowledge in koala conservation, citizen science and field activities. Strengthen cross-cultural and inter-generational knowledge exchange and develop partnerships for the management and conservation of koalas.	Commonwealth, state and territory government agencies in coordination with Indigenous land-owners, joint management partners and Indigenous ranger teams supported by natural resource management organisations, the National koala Recovery Team, and NGOs and the Zoo and Aquarium Association.	Ongoing (1)	TBA
2 e	Implement a comprehensive communication strategy for the plan's realisation.	Commonwealth, state and territory natural agencies and National koala Recovery Team; behavioural scientists	Ongoing (1)	TBA
2 f	Collaborate with existing database infrastructure to develop a user-friendly single-site portal for the general public to report koala sightings, together with awareness raising and encouragement; embed processes for regular updates and regular communication of information generated from the data.	Coordinated by the Commonwealth with state and territory government agencies; local NRM organisations and local government; NGOs and the Zoo and Aquarium Association.	Years 2-5 (1)	TBA

Action No.	Description	Potential Partners/Responsibility	Timeframe (Priority)	Est. cost
2 g	Build on existing guidance information with experts to develop national guidelines for veterinary standards in care, injuries, fertility control, disease treatment, tissue sampling, orphans and release for veterinarians, carers and koala rehabilitation centres; update and review to incorporate new learnings and knowledge.	Coordinated by the Commonwealth with state and territory government agencies, with input from research & veterinary experts; Expert Technical Advisory Panel; National Recovery Team; RSPCA and koala welfare organisations, including the Zoo and Aquarium Association.	Years 2-5 (2)	TBA
2 h	Implement community education and engagement programs in urban and peri-urban areas where impacts on koalas are high, incorporating best-practise understanding of values and attitudes towards koalas, responsible dog ownership and vehicle collisions and other urban issues resulting in koala deaths. These include, and are not limited to, developing and trialling innovative programs in koala aversion by dogs with owners; population and disease awareness; and reporting koala sightings.	State and territory government agencies in coordination with local government, traffic authorities and natural management organisations, welfare organisations, including the Zoo and Aquarium Association, and behavioural scientists; dog training organisations; RSPCA	Ongoing (1)	TBA

Strategy 3: Increase habitat protection

Land-use change is the most significant threat to the koala through habitat loss, fragmentation and degradation. Increasing the total area of protected, connected quality koala habitat in priority areas will be important to protect and recover koala populations. As koalas occur across different land tenures, notably private land, this will require a range of incentive mechanisms, including direct land purchases. Improvements in land management practices can also increase habitat protection without changing land use. While direct habitat protection forms some actions, this strategy primarily consists of developing incentives for such protection and thus this strategy has been included as a supporting strategy.

Action No.	Description	Potential Partners/Responsibility	Timeframe (Priority)	Est. cost
3 a	Increase the overall area of protected koala habitat by dedication of Crown land and purchasing land identified as priority koala habitat for incorporation into the state protected areas. Priority areas include those that support viable populations and those that have the greatest potential for population-level recovery.	States; territories; philanthropic investment	Ongoing (1)	TBA
3 b	Establish or expand existing targeted private land incentive mechanisms to increase the area for long-term protection and conservation of areas identified as priority koala habitats. Participation includes, but is not limited to, graziers, agricultural landholders, rural landholders, Indigenous land owners and private forestry.	States; territories; Commonwealth; philanthropic investment and NGOs. Indigenous land-owners and managers	Ongoing (2)	TBA
3 c	Improve the condition of existing koala habitat on both private and public land through altered land management practices, including management of vegetation, fire, weed, and introduced species.	State and territory government agencies; non-government land-owners; NGOs	Ongoing (2)	TBA
3 d	Investigate the potential to increase the protection of priority koala habitat through identification and registration of Critical Habitat where appropriate (i.e., Commonwealth-owned lands).	Commonwealth Government agencies; with strategic input from state and territory government agencies	Years 2-5 (2)	TBA

Strategy 4: koala conservation is integrated into policy, and statutory and land-use plans.

Management actions alone will not be sufficient to recover the koala. Actions are needed to ensure harmonisation of existing and future planning and policy settings such that they collectively contribute appropriately to maximising the chances of long-term survival of koalas in the wild.

Action No.	Description	Potential Partners/Responsibility	Timeframe (Priority)	Est. cost
4 a	Review and update EPBC Act referral guidelines for the listed koala to support regulatory decision making.	Commonwealth in consultation with state and territory governments, experts, planners, industry and the wider community,	Yr 1 (1)	$50,000
4 b	Review, revise, and strengthen where appropriate, statutory planning instruments, policies, and compliance controls at all levels of government, including local government, to avoid or minimise impacts of land use or land management on koala conservation. Embed principals of landscape-scale management.	State and territory government agencies in coordination with local government authorities; Commonwealth.	Yr 1 and ongoing (1)	TBA
4 c	Ensure identification and implementation of any offset decisions are strategic, coordinated, tracked in governments' databases, and informed by relevant planning and mapping documents such as NRM regional plans, Indigenous Heathy Country Plans associated with Indigenous Protected Areas (IPAs) or local government koala strategies.	Commonwealth, state and territory government agencies in coordination with local governments; National Recovery Team, Indigenous IPA managers	Yr 1 and ongoing (1)	TBA
4 d	Incorporate the impacts of climate change such as drought, heatwave and fire, into all strategic koala planning and actions, including restoration guidelines, offsets, translocation guidelines, forestry practices, corridor, reserve and protected area planning, allowing for iterative updates using a robust scenario-based approach	Commonwealth, state and territory government agencies in coordination with local governments	5 yearly (2)	TBA
4 e	Build on existing information to develop national guidelines or standards for koala-friendly urban design.	Commonwealth to coordinate state and territory government agencies, in consultation with local governments; urban planners	Yr 1 and ongoing (1)	TBA

Strategy 5: Strategic habitat restoration

Restoration increases the overall habitat available for koalas and increases the connectivity between areas of habitat, which is important to the long-term survival of koala populations. Many landcare-type organisations are restoring lost and degraded habitat for many species or to improve environmental functions. These activities are to ensure that resources are targeted to the most strategic areas.

Action No.	Description	Potential Partners/Responsibility	Timeframe (Priority)	Est. cost
5 a	Build on and implement landscape-scaled habitat restoration plans, including NRM regional plans, based on up-to-date mapping and spatial analysis that considers potential carrying capacity and landscape-scale	Coordinated approach between states and territory government agencies; local government; natural resource management agencies; NGOs	Ongoing (1)	TBA

	processes such as climate change, fire and drought, and koala movement patterns.			
5 b	Develop and implement best practice revegetation and restoration guidelines appropriate to local conditions that include planning for drought, heatwave, fire, and eucalypt responses to climate change using a robust scenario-based approach, consistent with national standards for ecological restoration (SERA 2017)	Coordinated between state and territory government agencies with input from research experts; Expert Technical Advisory Panel; natural resource management agencies and local community groups; NGOs	Years 1-5 (2)	TBA
5 c	Implement on-ground revegetation or restoration programs, following local restoration guidelines for the koala where they exist (e.g., NSW koala habitat revegetation guidelines (Wegner and Taws 2019)), in consultation with experts in koala ecology and plant geneticists. These should include experimental trialling of the establishment of climate resilient and nutritious feeding trees outside traditional ranges of koala habitat trees.	Coordinated approach between states and territory government agencies; local government; natural resource management agencies; local community groups; and NGOs.	Years 1-5 (2)	TBA

Strategy 6: Active metapopulation management

Metapopulation management concerns the movement of individuals and genes between populations. It is a complex and multi-faceted discipline. Adaptive management is the core of metapopulation management excellence. It requires consideration of cross-tenure land management, fire planning and operations, understanding of koala movement patterns and behavioural ecology, genetics, infection and disease, and fine-scale and macro-scale habitat needs, among other factors. To complicate these actions, planning instruments (e.g., development zoning) and forest harvesting practices are spatially variable, making it difficult to be prescriptive.

This strategy relies heavily on relevant and up-to-date habitat and distribution mapping and modelling for spatial prioritisation, climate change modelling, principles of landscape processes, and research into koala disease, population genetics habitat requirements, movement patterns, and biology. Management of fire, forest harvesting, and human activities and developments all influence koala metapopulations processes and must be managed to mitigate adverse impacts.

Action No.	Description	Potential Partners/Responsibility	Timeframe (Priority)	Est. cost
6 a	Develop meaningful and measurable metrics of health, genetics, population and distribution, at relevant planning scales, with triggers for management response. Integrate these triggers into metapopulation management, decision-making and programs. Implement response plans.	Commonwealth, state and territory government agencies, with input from research experts; National koala Monitoring Program; Expert Technical Advisory Panel and National Recovery Team	Years 1-5 (1)	TBA

Action No.	Description	Potential Partners/Responsibility	Timeframe (Priority)	Est. cost
6 b	Develop or build on existing best-practice koala translocation and post-care release guidelines for wild and captive populations, ensuring they are fit-for-purpose, informed by the latest research in metapopulation processes, genetics, disease and gut flora. Ensure the translocation guidelines are reviewed and updated within the life of this plan to integrate new understandings. If translocations are required, implement koala translocations in accordance with an appropriate decision framework and national guidelines (Wildlife Health Australia 2020), legislative requirements and consistent with international standards (IUCN/SSC 2013).	Coordinated by the Commonwealth with state and territory government agencies, with input from research experts; Expert Technical Advisory Panel & National Recovery Team; koala welfare organisations, including the Zoo and Aquarium Association, and RSPCA	5 yearly (2)	TBA
6 c	Regionally assess the feasibility, risks and cost-effectiveness of fire management options that seek to deliver long-term, strategic and landscape scale enhancement of the extent, and quality of current and future suitable habitat across tenures.	State and territory agencies with input from fire research experts; Expert Technical Advisory Panel and National Recovery Team; local fire authorities and local government, local landowners, Indigenous fire management practitioners & land-owners	Years 1-5 (1)	TBA
6 d	Develop and implement fire management that effectively secures and promotes long-term, strategic and effective protection of known populations and suitable habitat.	State and territory agencies with input from fire research experts; Expert Technical Advisory Panel and National Recovery Team; local fire authorities and local government; koala welfare organisations and RSPCA	Years 1-5 (1)	TBA
6 e	Develop and implement response and decision-support tools for individual and population management in emergencies such as bushfire, drought and floods. These include support and coordination of carer networks.	Coordinated by the Commonwealth with state and territory government resource agencies, local government agencies, natural resource management agencies and koala welfare organisations, with input from research experts; Expert Technical Advisory Panel and National Recovery Team	Years 1-5 (1)	TBA

Recovery plan decision

A decision has been made to have a Recovery Plan due to the 2012 recommendation by the Threatened Species Scientific Committee (TSSC 2012). This recovery plan is currently being drafted in parallel with this document.

Links to relevant implementation documents

Species Profile and Threats Database: http://www.environment.gov.au/cgi-bin/sprat/public/publicspecies.pl?taxon_id=85104

Revised provisional list of animals requiring urgent management intervention following the 2019-2020 bushfires: https://www.environment.gov.au/biodiversity/bushfire-recovery/priority-animals

NSW koala Strategy: https://www.environment.nsw.gov.au/topics/animals-and-plants/threatened-species/programs-legislation-and-framework/nsw-koala-strategy

Saving our species Framework for the spatial prioritisation of koala conservation actions in NSW Iconic koala Project. https://www.environment.nsw.gov.au/research-and-publications/publications-search/framework-for-the-spatial-prioritisation-of-koala-conservation-actions-in-nsw

South East Queensland Koala Conservation Strategy: https://environment.des.Qld.gov.au/wildlife/animals/living-with/koalas/conservation/seq-koala-strategy

Advice to the Minister for Sustainability, Environment, Water, Population and Communities from the Threatened Species Scientific Committee (the Committee) on Amendment to the list of Threatened Species under the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act): http://www.environment.gov.au/biodiversity/threatened/species/pubs/197-listing-advice.pdf

Conservation Advice and Listing Assessment references

Abram NJ, Henley BJ, Gupta AS, Lippmann TJ, Clarke H, Dowdy AJ, Sharples JJ, Nolan RH, Zhang T & Wooster MJ (2021). Connections of climate change and variability to large and extreme forest fires in southeast Australia. *Communications Earth & Environment 2*,1-17.

Adams-Hosking C, Grantham HS, Rhodes JR, McAlpine C & Moss PT (2011). Modelling climate-change-induced shifts in the distribution of the koala. *Wildlife Research 38*,122-130.

Adams-Hosking C, McBride MF, Baxter G, Burgman M, De Villiers D, Kavanagh R, Lawler I, Lunney D, Melzer A & Menkhorst P (2016). Use of expert knowledge to elicit population trends for the koala (Phascolarctos cinereus). *Diversity and Distributions 22*,249-262.

Au J, Clark RG, Allen C, Marsh KJ, Foley WJ & Youngentob KN (2019). A nutritional mechanism underpinning folivore occurrence in disturbed forests. *Forest Ecology and Management 453*,117585.

Ballantyne K, Lisle A, Mucci A & Johnston SD (2015). Seasonal oestrous cycle activity of captive female koalas in south-east Queensland. *Australian Mammalogy 37*,245-252.

BoM (2019). Special Climate Statement 68—widespread heatwaves during December 2018 and January 2019. http://www.bom.gov.au/climate/current/statements/scs68.pdf.

BoM (2021a). Climate Change in Australia. Projections for Australia's NRM regions. Bureau of Meteorology. Accessed: 15 February 2021 https://www.climatechangeinaustralia.gov.au/en/climate-projections/future-climate/regional-climate-change-explorer/super-clusters/?current=ESC&tooltip=true&popup=true.

BoM (2021b). Heatwave Assessment and Forecast. Bureau of Meteorology. Accessed: 15 February 2021 http://www.bom.gov.au/metadata/catalogue/19115/ANZCW0503900601.

Bradshaw S, Dixon K, Lambers H, Cross A, Bailey J & Hopper S (2018). Understanding the long-term impact of prescribed burning in mediterranean-climate biodiversity hotspots, with a focus on south-western Australia. *International Journal of Wildland Fire 27*,643-657.

Cahir F, Schlagloth R & Clark ID (2021). The Importance of the Koala in Aboriginal Society in Nineteenth-Century Queensland (Australia): A Reconsideration of the Archival Record. *Anthrozoös*,1-15.

Cahir F, Sclagoth R & Clark ID (2020). The Historic Importance of the Koala in Aboriginal Society in New South Wales, Australia: An Exploration of the Archival Record. *ab-Original 3*,172-191.

Capital-Ecology (2018). Koala Phascolarctos cinereus surveys in the Australian Capital Territory, 2018. A Report to the Environment, Planning and Sustainable Development Directorate, ACT Government, Canberra. https://www.environment.act.gov.au/__data/assets/pdf_file/0020/1255142/Koala-Survey-Report-Web-Accessible.pdf.

Clark G & Johnston E (2016). Coasts: Population growth and urban development: Coastal development and land use.', in Australia State of the Environment 2016, Australian Government Department of the Environment and Energy (former), Canberra.

Climate-Council (2019). Climate Council Briefing Paper: 'This is Not Normal': Climatechange and escalating bushfire risk. https://www.climatecouncil.org.au/wp-content/uploads/2019/11/CC-nov-Bushfire-briefing-paper.pdf.

Davies NA, Gramotnev G, McAlpine C, Seabrook L, Baxter G, Lunney D, Rhodes JR & Bradley A (2013). Physiological Stress in Koala Populations near the Arid Edge of Their Distribution. PLOS ONE 8,e79136.

DAWE (2020) Area of Occupancy and Extent of Occurrence for *Phascolarctos cinereus*. Unpublished report. Department of the Environment (Commonwealth), Canberra.

DAWE (2021a). Bushfire Recovery Environmental Analysis Decision Support (BREADS) tool. V21_18_IBRA. Department of Agriculture, Water and Environment.

DAWE (2021b). National Koala Monitoring Workshop, February 1-2 (2021). Department of Agriculture, Water and Environment. In. Virtual workshop, hosted by DAWE, Canberra, Australia

DeGabriel JL, Moore BD, Foley WJ & Johnson CN (2009). The effects of plant defensive chemistry on nutrient availability predict reproductive success in a mammal. *Ecology 90*,711-719.

DES (2018). Land cover change in Queensland: a Statewide Landcover and Trees Study Summary Report: 2016–17 and 2017–18, Queensland Department of Environment and Science, Brisbane, available from https://www.qld.gov.au/__data/assets/pdf_file/0031/91876/landcover-change-in-queensland-2016-17-and-2017-18.pdf.

DES (2020). South East Queensland Koala Conservation Strategy 2020–2025. https://environment.des.qld.gov.au/__data/assets/pdf_file/0016/211732/seq-koala-conservation-strategy-2020-2025.pdf.

DEW (2018). A Review of the Evidence for the Management of South Australia's Free-ranging Koala Populations. DEW Technical note 2018/11, Government of South Australia, Department for Environment and Water, Adelaide.

Di Virgilio G, Evans JP, Blake SA, Armstrong M, Dowdy AJ, Sharples J & McRae R (2019). Climate change increases the potential for extreme wildfires. *Geophysical Research Letters 46*,8517-8526.

Dissanayake R, Stevenson M, Allavena R & Henning J (2021). Predicting koala (Phascolarctos cinereus) distribution from incidental sighting data in South-East Queensland, Australia. *Global Ecology and Conservation 28*,e01662.

DofE (2014). EPBC Act referral guidelines for the vulnerable koala (combined populations of Queensland, New South Wales and the Australian Capital Territory), Commonwealth of Australia, 2014. https://www.environment.gov.au/system/files/resources/dc2ae592-ff25-4e2c-ada3-843e4dea1dae/files/koala-referral-guidelines.pdf.

DPIE (2020). Saving our species. Framework for the spatial prioritisation of koala conservation actions in NSW. Iconic Koala Project. State of NSW and Department of Planning, Industry and Environment. https://www.environment.nsw.gov.au/-/media/OEH/Corporate-Site/Documents/Animals-and-plants/Threatened-species/framework-spatial-prioritisation-koala-conservation-190045.pdf.

Eldridge M & Lott M (2020). NESP project (8.3.3) to assess spatial genetics and species boundaries for fire-affected vertebrates: Koala Phascolarctos cinereus. Provided by Craig Moritz (13-10-20).

Ellis W, Bercovitch F, FitzGibbon S, Melzer A, De Villiers D & Dique D (2010a). Koala birth seasonality and sex ratios across multiple sites in Queensland, Australia. *Journal of Mammalogy 91*,177-182.

Ellis W, Melzer A, Clifton I & Carrick F (2010b). Climate change and the koala Phascolarctos cinereus: water and energy. *Australian Zoologist 35*

Evans MC (2016). Deforestation in Australia: drivers, trends and policy responses. *Pacific Conservation Biology 22*,130-150.

Fitzgerald J (2014). Koala Sightings email sent to Wildcare on 5th June 2014. Email archived at: http://bioacoustics.cse.unsw.edu.au/archives/html/canberrabirds/2014-06/msg00034.html. Access date: May 7, 2021. In: Possumpost

Goldfuß A & Bischof G (1817). 1817, Physikalisch-statistische Beschreibung des Fichtelgebirges, Stein, Nürnberg.

Gonzalez-Astudillo V (2018). Analysis of morbidity and mortality of wild koalas in south-east Queensland using passive surveillance data. Thesis. The University of Queensland.

Heagney EC, Falster D & Kovač M (2021). Land clearing in south-eastern Australia: Drivers, policy effects and implications for the future. *Land Use Policy 102*,105-243.

Heard, GW and Ramsey, DSL (2020). Modelling Koala abundance across Victoria. Unpublished Client Report for Biodiversity Division, Department of Environment, Land, Water and Planning. Arthur Rylah Institute for Environmental Research, Department of Environment, Land, Water and Planning, Heidelberg, Victoria.

IUCN (2019). IUCN Standards and Petitions Committee. 2019. Guidelines for Using the IUCN Red List Categories and Criteria. Version 14. Prepared by the Standards and Petitions Committee. Downloadable from http://www.iucnredlist.org/documents/RedListGuidelines.pdf. In:

IUCN/SSC (2019). Guidelines for Using the IUCN Red List Categories and Criteria, Version 14, IUCN Species Survival Commission, Gland, Switzerland, available from https://www.iucnredlist.org/resources/redlistguidelines.

Johnson RN, O'Meally D, Chen Z, Etherington GJ, Ho SYW, Nash WJ, Grueber CE, Cheng Y, Whittington CM, Dennison S, Peel E, Haerty W, O'Neill RJ, Colgan D, Russell TL, Alquezar-Planas DE, Attenbrow V, Bragg JG, Brandies PA, Chong AY-Y, Deakin JE, Di Palma F, Duda Z, Eldridge MDB, Ewart KM, Hogg CJ, Frankham GJ, Georges A, Gillett AK, Govendir M, Greenwood AD, Hayakawa T, Helgen KM, Hobbs M, Holleley CE, Heider TN, Jones EA, King A, Madden D, Graves JAM, Morris KM, Neaves LE, Patel HR, Polkinghorne A, Renfree MB, Robin C, Salinas R, Tsangaras K, Waters PD, Waters SA, Wright B, Wilkins MR, Timms P & Belov K (2018). Adaptation and conservation insights from the koala genome. *Nature Genetics 50*,1102-1111.

Keith D, Akçakaya HR, Butchart SHM, Collen B, Dulvy NK, Holmes EE, Hutchings JA, Keinath D, Schwartz MK, Shelton AO & Waples RS (2015). Temporal correlations in population trends: conservation implications from time-series analysis of diverse animal taxa. *Biological Conservation 192*,247-257.

Kjeldsen SR, Raadsma HW, Leigh KA, Tobey JR, Phalen D, Krockenberger A, Ellis WA, Hynes E, Higgins DP & Zenger KR (2019). Genomic comparisons reveal biogeographic and

anthropogenic impacts in the koala (Phascolarctos cinereus): a dietary-specialist species distributed across heterogeneous environments. *Heredity 122*,525-544.
Leavesley A, Wuters M & Thornton R (2020). Prescribed burning in Australasia. The science, practice and politics of burning the bush. . Australasian Fire and Emergency Service Authorities Council Limited, Melborurne, Victoria.
Legge S, Woinarski JCZ, Garnett ST, Geyle H, Lintermans M, Nimmo DG, Rumpff L, Scheele BC, Southwell DG, Ward M, Whiterod NS, Ahyong S, Blackmore C, Bower D, Brizuela Torres D, Burbidge AH, Burns P, Butler G, Catullo R, Dickman CR, Doyle K, Ensby, M., , Ehmke G, Ferris J, Fisher D, Gallagher R, Gillespie G, Greenlees MJ, Hayward-Brown B, Hohnen R, Hoskin C, Hunter D, Jolly C, Kennard M, King A, Kuchinke D, Law B, Lawler I, Loyn R, Lunney D, Lyon J, MacHunter J, Mahony M, Mahony S, McCormack R, Melville J, Menkhorst P, Michael D, Mitchell N, Mulder E, Newell D, Pearce L, Raadik T, Rowley J, Sitters H, Spencer R, Lawler S, Valavi R, Ward M, West M, Wilkinson D & Zukowski S (2021). Estimates of the impacts of the 2019-2020 fires on populations of native animal species. NESP Threatened Species Recovery Hub project 8.3.2 report. (Brisbane, Australia.).
Lunney D, Crowther MS, Shannon I & Bryant JV (2009). Combining a map-based public survey with an estimation of site occupancy to determine the recent and changing distribution of the koala in New South Wales. *Wildlife Research 36*,262-273.
Lunney D, Crowther MS, Wallis I, Foley WJ, Lemon J, Wheeler R, Madani G, Orscheg C, Griffith JE & Krockenberger M (2012). Koalas and climate change: a case study on the Liverpool Plains, north-west New South Wales. *Wildlife and Climate Change: towards robust conservation strategies for Australian Fauna'.(Eds D. Lunney and P. Hutchings.) pp*,150-168.
Lunney D, Stalenberg E, Santika T & Rhodes JR (2014). Extinction in Eden: identifying the role of climate change in the decline of the koala in south-eastern NSW. *Wildlife Research 41*,22-34.
Markwell K (2020a). Getting close to a national icon: an examination of the involvement of the koala (Phascolarctos cinereus) in Australian tourism. *Tourism Recreation Research*,1-14.
Markwell K (2020b). Koalas are the face of Australian tourism. What now after the fires? https://theconversation.com/koalas-are-the-face-of-australian-tourism-what-now-after-the-fires-129347. *The Conversation*
Martin R & Handasyde KA (1999). The koala: natural history, conservation and management. UNSW press
McAlpine C, Lunney D, Melzer A, Menkhorst P, Phillips S, Phalen D, Ellis W, Foley W, Baxter G, de Villiers D, Kavanagh R, Adams-Hosking C, Todd C, Whisson D, Molsher R, Walter M, Lawler I & Close R (2015). Conserving koalas: A review of the contrasting regional trends, outlooks and policy challenges. *Biological Conservation 192*,226-236.
McLean N & Handasyde KA (2007). Sexual maturity, factors affecting the breeding season and breeding in consecutive seasons in populations of overabundant Victorian koalas (Phascolarctos cinereus). *Australian Journal of Zoology 54*,385-392.
Melzer A, Carrick F, Menkhorst P, Lunney D & John BS (2000). Overview, critical assessment, and conservation implications of koala distribution and abundance. *Conservation Biology 14*,619-628.
Melzer A, Santamaria F & Allen S (2018). The koalas, koala habitat and conservation management in the Clarke-Connors Ranges and associated landscapes. A report to the Queensland Department of Transport and Main Roads. Koala Research CQ, School of Medical and Applied Sciences, CQUniversity, Rockhampton.
Melzer A & Tucker G (2011). Koalas of the St Lawrence region of Central Queensland. Report 1: Defining the population. This report was prepared for the Queensland Department of Transport and Main Roads.
Menkhorst P (2008). Menkhorst, P 2008, 'Hunted, marooned, re-introduced, and contracepted: A history of Koala management in Victoria' in D Lunney, L Munn and W Meikle (eds), Too

Close for Comfort: Contentious Issues in Human-wildlife Encounters, Royal Zoological Society of NSW, Mosman, NSW.,73-92.
Moore BD & Foley WJ (2005). Tree use by koalas in a chemically complex landscape. *Nature 435*,488-490.
Moore BD, Lawler IR, Wallis IR, Beale CM & Foley WJ (2010). Palatability mapping: a koala's eye view of spatial variation in habitat quality. *Ecology 91*,3165-3176.
Munks S, Corkrey R & Foley W (1996). Characteristics of arboreal marsupial habitat in the semi-arid woodlands of northern Queensland. *Wildlife Research 23*,185-195.
Narayan E (2019). Physiological stress levels in wild koala sub-populations facing anthropogenic induced environmental trauma and disease. *Scientific Reports 9*,6031.
NOAA (2021). Climate Model: Temperature Change (RCP 6.0) - 2006 - 2100. https://sos.noaa.gov/datasets/climate-model-temperature-change-rcp-60-2006-2100/. Accessed 2020. In:
NSW Government (2014). Adapt NSW - New England north west climate change snapshot. Office of Environment and Heritage. .
NSW Government (2019a). Koala Habitat Information Base. Created by Department of Planning, Industry and Environment. https://www.environment.nsw.gov.au/topics/animals-and-plants/threatened-species/programs-legislation-and-framework/nsw-koala-strategy/building-knowledge-on-koala-habitat. In:
NSW Government (2019b). NSW Koala Likelihood Map v2.0 (August 2019). Department of Planning, Industry and Environment. https://researchdata.edu.au/nsw-koala-likelihood-august-2019/1426089.
NSW Government (2020). Saving our species Framework for the spatial prioritisation of koala conservation actions in NSW Iconic Koala Project. https://www.environment.nsw.gov.au/research-and-publications/publications-search/framework-for-the-spatial-prioritisation-of-koala-conservation-actions-in-nsw.
NSW Government (2021a). Koala research in NSW forests. NSW Government, Department of Primary Industries. https://www.dpi.nsw.gov.au/forestry/science/koala-research.
NSW Government (2021b). Monitoring Koalas in Hinterland Forests of Northeast NSW and the effect of 2019 fires on the meta-population. Department of Primary Industries. https://www.dpi.nsw.gov.au/forestry/science/koala-research.
Pacifici M, Santini L, Di Marco M, Baisero D, Francucci L, Grottolo Marasini G, Visconti P & Rondinini C (2013). Generation length for mammals. *Nature Conservation 5*,87-94.
Phillips SS (2000). Population trends and the koala conservation debate. *Conservation Biology 14*,650-659.
Pokhrel Y, Felfelani F, Satoh Y, Boulange J, Burek P, Gädeke A, Gerten D, Gosling SN, Grillakis M & Gudmundsson L (2021). Global terrestrial water storage and drought severity under climate change. *Nature Climate Change*,1-8.
Polkinghorne A, Hanger J & Timms P (2013). Recent advances in understanding the biology, epidemiology and control of chlamydial infections in koalas. *Veterinary microbiology 165*,214-223.
Predavec M, Lunney D, Shannon I, Lemon J, Sonawane I & Crowther M (2018). Using repeat citizen science surveys of koalas to assess their population trend in the north-west of New South Wales: scale matters. *Australian Mammalogy 40*,47-57.
Queensland-Government (2021). Koala hospital data extracted from the Department of Environment and Science koala hospital database (KoalaBase) between July 1996 - December 2019. Viewed 19/3/21. https://www.data.qld.gov.au/dataset/koala-hospital-data/resource/7c6f7da8-ef7a-48e4-bf4e-c449a885e46d.
Rennison B & Fisher M (2018). Framework for the spatial prioritisation of koala conservation actions in NSW Saving our Species Iconic Koala Project. A report prepared for the Office of Environment and Heritage. 2018 State of NSW and Office of Environment and Heritage.

Robinson EA, Ryan GD & Newman JA (2012). A meta-analytical review of the effects of elevated CO2 on plant–arthropod interactions highlights the importance of interacting environmental and biological variables. *New Phytologist 194*,321-336.

Runge C, Rhodes J & Latch P (2021a). A national approach to the integration of koala spatial data to inform conservation planning, NESP Threatened Species Recovery Hub Project 4.4.12 report. The University of Queensland, Brisbane, Spatial data and supporting documents available at https://doi.org/10.5281/zenodo.4305157.

Runge CA, Rhodes J.R & Lopez-Cubillos DS (2021b). Harmonised koala habitat mapping. Version 1.0. NESP Threatened Species Recovery Hub Project 4.4.12 report. The University of Queensland, Brisbane. https://doi.org/10.5281/zenodo.4305167.

Santika T, McAlpine CA, Lunney D, Wilson KA & Rhodes JR (2014). Modelling species distributional shifts across broad spatial extents by linking dynamic occupancy models with public-based surveys. *Diversity and Distributions 20*,786-796.

Santika T, McAlpine CA, Lunney D, Wilson KA & Rhodes JR (2015). Assessing spatio-temporal priorities for species' recovery in broad-scale dynamic landscapes. *Journal of Applied Ecology 52*,832-840.

Schlagloth R, Cahir F & Clark I (2018). The Importance of the Koala in Aboriginal Society in Nineteenth-century Victoria (Australia): A Reconsideration of the Archival Record. *Anthrozoös 31*,433-441.

Seabrook L, McAlpine C, Baxter G, Rhodes J, Bradley A & Lunney D (2011). Drought-driven change in wildlife distribution and numbers: a case study of koalas in south west Queensland. *Wildlife Research 38*,509-524.

SEWPaC (2012a). Approved Conservation Advice for Phascolarctos cinereus (combined populations in Queensland, New South Wales and the Australian Capital Territory). Canberra: Department of Sustainability, Environment, Water, Population and Communities. http://www.environment.gov.au/biodiversity/threatened/species/pubs/197-conservation-advice.pdf. In effect under the EPBC Act from 02-May-2012.

SEWPaC (2012b). Federal Register of Legislation. Determination that a distinct population of biological entities is a species for the purposes of the Environment Protection and Biodiversity Conservation Act 1999 (Cth) (132). Department of Sustainability, Environment, Water, Population and Communities. https://www.legislation.gov.au/Details/F2012L00960.

Sherwin WB, Timms P, Wilcken J & Houlden B (2000). Analysis and Conservation Implications of Koala Genetics. *Conservation Biology 14*,639-649.

Simmons BA, Wilson KA & Dean AJ (2021). Psychosocial drivers of land management behaviour: How threats, norms, and context influence deforestation intentions. https://doi.org/10.1007/s13280-020-01491-w *Ambio*

Stalenberg E, Wallis IR, Cunningham RB, Allen C & Foley WJ (2014). Nutritional correlates of koala persistence in a low-density population. *PLoS One 9*,e113930.

TSSC (2012). Koala (*Phascolarctos cinereus*) Listing (2012). http://www.environment.gov.au/biodiversity/threatened/species/pubs/197-listing-advice.pdf. In effect under the EPBC Act from 02-May-2012.

Tucker G, Melzer A & Ellis W (2008). The development of habitat selection by subadult koalas. *Australian Journal of Zoology 55*,285-289.

Ward MS, Simmonds JS, Reside AE, Watson JEM, Rhodes JR, Possingham HP, Trezise J, Fletcher R, File L & Taylor M (2019). Lots of loss with little scrutiny: The attrition of habitat critical for threatened species in Australia. *Conservation Science and Practice 1*,e117.

Wedrowicz F, Wright W, Schlagloth R, Santamaria F & Cahir F (2017). Landscape, koalas and people: A historical account of koala populations and their environment in South Gippsland. *Australian Zoologist 38*,518-536.

Wilmott L (2020). Talk: Koala home range size and chlamydia disease expression correlated with soil fertility. Ecological Society of Australia conference 2020. https://www.esa2020.org.au/full-scientific-program/. In:

Woinarski JB, A.A. (2020). Phascolarctos cinereus (amended version of 2016 assessment). The IUCN Red List of Threatened Species 2020: e.T16892A166496779. https://dx.doi.org/10.2305/IUCN.UK.2020-1.RLTS.T16892A166496779.en. Downloaded on 13 October 2020.

Woinarski JCZ, Burbidge AA & Harrison PL (2014). The Action Plan for Australian Mammals 2012. CSIRO publishing, Collingwood.

Wujeska-Klause A, Crous KY, Ghannoum O & Ellsworth DS (2019). Lower photorespiration in elevated CO2 reduces leaf N concentrations in mature Eucalyptus trees in the field. *Global change biology 25*,1282-1295.

The Threatened Species Scientific Committee finalised this assessment on 07 September 2021.

Attachment A: Listing Assessment for *Phascolarctos cinereus* combined populations of Queensland, New South Wales and the Australian Capital Territory

Reason for assessment

This assessment follows prioritisation of a nomination from the TSSC, initiated in response to the 2019/20 fires.

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	92,184	86,863	92,184	**Past population data for the listed koala:** **2001** population estimate used in calculations: 184,7400 Data hindcast from the 2012 expert elicitation (Adams-Hosking et al. 2016). **2012** population estimate: 115,600 Data source: 2012 expert elicitation (Adams-Hosking et al. 2016). **2021** population estimate: 92,200. **2032** population estimate: 63,500. Data sources: 2012 expert elicitation (Adams-Hosking et al. 2016) and 2021 expert elicitation (Legge et al. 2021)
Trend	contracting			
Generation time (years)	6.5 years	6 years	7 years	Using conservative values of sexual maturity at 3 years and longevity 15 years, generation time is estimated to be approximately 6.5 years. Here the three generation period is considered to be 20 years.

Extent of occurrence	1,665,850 km²			Data provided by Department of Agriculture Water and Environment, Geoscience Australia and PSMA Australia.
Trend	contracting			
Area of Occupancy	19,428 km²			The area of occupancy is estimated at 19,400 km². These figures are based on the mapping of point records from 2000 from state governments, museums and CSIRO. Due to the lack of recent surveys more recent data cannot be used to predict range contraction.
Trend	contracting Contracting due to climate related threats and habitat loss and land clearance.			
Number of subpopulations	>10			Geographically isolated populations exist throughout the koala's range due to habitat fragmentation resulting from large scale land clearing, drought and bushfire impacts. Populations West of the Great Dividing Range are considered to be isolated from their eastern counterparts (DAWE 2021b). Koala habitat is patchy and fragmented and increasingly prone to threats from drought resulting in multiple subpopulations (n≥3). In, Queensland, koala populations to the north (e.g., Wet Tropics), western inland arid regions (e.g., Mulga Lands) and southern end of the state (e.g., South East Queensland) are increasingly isolated due to habitat loss and fragmentation (DES 2020) (n≥3). In New South Wales, the east coast was heavily impacted by 2019-2020 bushfires. While the extent of bushfires was large, the fire intensity varied from low to high. Ongoing research indicates that areas of high intensity fire have zero koala occupancy in 2021. In contrast, low severity and moderate severity fire impacted areas are reported to have 100% koala occupancy (Pers comm., Natural Resources Commission 2021 koala Annual Forum). The high intensity fire impacts are likely to have the worst impact in poorly connected subpopulations (n≥5). Preliminary genetic analysis also confirms that there is no longer genetic exchange across the Clarence River in NSW, or from the north to

				the south of the Sydney basin (Eldridge & Lott 2020).
Trend	Declining The number of subpopulations is declining as climate suitable koala habitat shrinks.			
Basis of assessment of subpopulation number	The number of koala subpopulations is based on the available data and barriers to connectivity.			
No. locations	>10			
Trend	unknown			
Basis of assessment of location number	The spatial nature of the threats, although stochastic in time and space, is such that there are > 10 geographically or ecologically distinct areas where a single threatening event (e.g., drought or fire) could affect all of the individuals present within a single generation. The geographic location of non-impacted locations will vary between events, but there are always likely to be > 10.			
Fragmentation	Increasingly fragmented–e.g., by the 2019/20 fires.			
Fluctuations	Data deficient.			

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 A2c, A4c **for listing as** Endangered

For the listed koala (Queensland, New South Wales and the Australian Capital Territory):

Generation length

Female koalas reach sexual maturity between 2 and 3 years of age (McLean & Handasyde 2007). In the wild, longevity can be more than fifteen years for females and more than twelve years for males (Martin & Handasyde 1999). IUCN Guidelines (2019) provide the following as one method for estimation of generation length:

> Age of first reproduction + [z * (length of the reproductive period)], where z is a number between 0 and 1

For mammals, values of z have been estimated at 0.29 and 0.284 (Pacifici et al. 2013; Keith et al. 2015).

Using conservative values of sexual maturity at 3 years and longevity 15 years, generation time is estimated to be approximately 6.5 years. Here the three generation period is considered to be 20 years.

Evidence - estimated

A2 Past population reductions (2001-2021):

The total number of koalas in Queensland, New South Wales, and the Australian Capital Territory in the year 2001 – the starting point for this assessment – was estimated to be between 184,748 and 170,335. This estimate was derived from bioregional population estimates for 2012 provided by Adams-Hosking et al. (2016). These bioregional estimates sum to a total population of 115,614 in 2012 (Adams-Hosking et al. 2016); a figure that is widely accepted by state governments, non-government organisations (NGOs) and researchers and builds on the 2012 EPBC listing advice (TSSC 2012). The 2012 bioregional population estimates were adjusted by Adams-Hosking et al.'s (2016) estimates of the rate of decline in each bioregion over the preceding three generations to yield bioregional population estimates for the year 1992. We then derived bioregional values for the year 2001 by assuming that the form of decline in each bioregion between 1992 and 2012 was either linear (giving the summed estimate of 184,748) or exponential (170,335); note that Adams-Hosking et al. (2016) did not specify the shape of the decline curve over the three-generation period. Total population estimates for the year 2021 were derived similarly, by projecting the bioregional declines from 1992-2012 forward to 2021. The resulting values for the total population in 2021 were 92,184 (linear decline) and 86,863 (exponential decline) (Table 4, Box 1).

Table 4 shows that that for the period 2001 to 2021 the estimated decline of the total population reaches the Endangered threshold of 50 percent under this Criterion. Whether the shape of the decline curve is exponential or linear has little effect on the outcome. Key bioregions (e.g., Mulga Lands) likely did not decline in a linear or exponential fashion, but rather were relatively stable until around 2000 then declined precipitously due to the Millennium Drought (Seabrook et al. 2011). If this "step change" were factored into the calculations in Table 4 it would have the effect of estimating a higher population at the beginning of the assessment period for Criterion A2, and thus a proportionally higher rate of decline.

Additionally, these data do not include the effects of the 2019/20 bushfires. While fire was considered as a threat in the elicitation exercise of Adams-Hosking et al. (2016), fires of the scale of 2019/20 were not anticipated in estimating declines that were likely to occur after 2012 (Hosking, Kavanagh, Lawler, Lunney, Melzer, Menkhorst, Moore pers comm April 2021). Thus again, this analysis likely underestimates the overall decline.

In a project run by the Threatened Species Recovery Hub in 2020-21, expert elicitation was used to estimate the likely mortality of koalas in low/med and high/very high severity fires. These estimates were then combined with spatial estimates of the proportion of the listed koala's range that was burned in those severity classes, to estimate the overall population reduction caused by the fire. It was estimated that populations declined by 10 percent (80 percent confidence 5.0 to 17 percent) by one year after 2019/20 fires, and that they would continue to decline thereafter without returning to their pre-fire population size. This analysis assumed uniform density of koalas across their range. However, the fires occurred predominantly in areas where koala densities are relatively higher than, for example, in large parts of their range west of the Great Divide, and thus this estimate likely underestimates the mortality due to the fires.

The estimated decline sits on the lower threshold for the Endangered category. Thus, while the effects of the "step change" due to drought and the similar sudden drop in numbers due to the 2019/2020 fires cannot be accurately quantified, it can confidently be concluded that they move the estimate well into the Endangered range. They are unlikely to be of sufficient scale to reach the threshold for the Critically Endangered category, which would require an overall decline of ≥80 percent for this criterion (Table 4). Consequently, given that the koala is demonstrably close to the lower threshold of Endangered and that ongoing trends suggest further events likely to be sufficient to worsen the decline, the Committee considers that the koala is eligible for listing as Endangered under this subcriterion A2c.

Table 4 – Estimated population sizes for bioregions containing koalas, calculated from the values provided in an expert elicitation study estimating koala sizes and trends +/- three generations from 2012.

Values for 2032 generated directly by applying three generation trends. Values for 2001 estimated by hindcasting three generations back to 1992 then calculated based on assuming either constant linear, or exponential, decline across the three generation period. Values for 2021 also based on constant linear, or exponential, decline between 2012 and 2032. Full details of these calculations are shown in Box 1 for the Brigalow Belt North bioregion as an exemplar.

Bioregion	2012	Past or future change (%) over 3 gens	Hindcast (ca 1992)	2001 linear	2021 linear	2001 exponential	2021 exponential decline	Forecast (ca. 2032)
				A2				A4
Cobar Peneplain & Riverina	2,354	-9	2,587	2,482	2,259	2,480	2,256	2,142
Darling Riverine Plains	964	-34	1,461	1,237	816	1,212	800	636
Mulga Lands (NSW)	711	-31	1,030	886	612	872	602	491
Murray Darling Depression	55	-12	63	59	52	59	52	48
New England Tablelands	2,771	6	2,614	2,685	2,846	2,683	2,845	2,937
NSW North Coast	8,367	-50	16,734	12,969	6,485	12,250	6,125	4,184
NSW South Western Slopes	2,310	-23	3,000	2,690	2,071	2,667	2,054	1,779
South Brigalow & Nandewar	11,133	-35	17,128	14,430	9,379	14,110	9,171	7,236
South East Corner	655	-46	1,213	962	520	919	496	354
South Eastern Highlands	1,363	-19	1,683	1,539	1,246	1,531	1,240	1,104
Sydney Basin	5,667	-4	5,903	5,797	5,565	5,796	5,564	5,440
Brigalow Belt North	15,179	-63	41,024	29,394	10,876	26,226	9,704	5,616
Brigalow Belt South	11,071	-56	25,161	18,821	8,281	17,389	7,651	4,871
Central Mackay Coast	8,857	-35	13,626	11,480	7,462	11,225	7,296	5,757
Desert Uplands	6,357	-20	7,946	7,231	5,785	7,187	5,750	5,086
Einasleigh Uplands & Wet Tropics	4,750	-41	8,051	6,566	3,874	6,349	3,746	2,803
Mitchell Grass Downs	1,943	-39	3,185	2,626	1,602	2,550	1,556	1,185
Mulga Lands (QLD)	15,286	-73	56,615	38,017	10,264	31,408	8,480	4,127
South Eastern Queensland	15,821	-51	32,288	24,878	12,190	23,422	11,477	7,752
TOTAL	**115,614**		**241,312**	**184,748**	**92,184**	**170,335**	**86,863**	**63,549**
Estimated decline over three generations				**50%**		**49%**		**45%**

Box 1. Example of calculations used in calculating time-corrected estimates - Brigalow Belt North Bioregion

Notes:
1. Because the estimated declines rates vary between bioregions, the calculations were made for each bioregion and summed across the relevant area to provide overall estimates. One bioregion is shown here as an exemplar.
2. For simplicity, numbers used below are rounded, but this was not the case when calculations were made on a spreadsheet and thus it may appear that there are minor discrepancies with Table 4.

Adams-Hosking et al. (2016) estimated that in 2012 the population of koalas in this bioregion was 15,179 and that the decline over the past, and future, three generations from 2012 was 63 percent.

Hindcast to previous three generations from 2012 (i.e., approximately 1992)
$N_{2012} = N_{1992}*(100\%-63\%)$
$N_{2012} = N_{1992}*37\%$
$N_{2012}/37\% = N_{1992}$
$N_{1992} = 15{,}179/37\%$
$= 41{,}024$

Forecast to following three generations from 2012 (i.e., approximately 2032)
$N_{2032} = N_{2012}*(100\%-63\%)$
$= 15{,}179*.37$
$= 5{,}616$

Estimating population at beginning of relevant three generation time period for Criterion A2 (i.e., approx. 2001)
Assuming linear decline
$N_{2012} = 15{,}179$
$N_{1992} = 41{,}024$
Decline/year = $(N_{2012} - N_{1992})/(2012-1992)$
$= (41{,}024-15{,}179)/20$
$= 25{,}845/20$
$= 1292$
$N_{2001} = N_{1992}$-(Decline/year)*(2001-1992)
$= 41{,}024-(1292*9)$
$= 29{,}394$

Assuming exponential decline
$N_{1992} = 41{,}024$
Decline over 20 years = 63%
Remaining = 37% = 0.37
Decline/year = $0.37^{(1/20)} = 0.952$
$N_{2001} = N_{1992}*0.952^{(2001-1992)}$
$= N_{1992}*0.952^{9}$
$= 41{,}024*0.639$
$= 26{,}226$

Estimating population at end of relevant three generation time period for Criterion A2 (i.e., approx. 2021)

Assuming linear decline

N_{2012} = 15,179
N_{2032} = 5,616
Decline/year = (N_{2032} - N_{2012})/(2032-2012)
= (15,179-5,616)/20
= 9,563/20
= 478
N_{2021} = N_{2012}-(Decline/year)*(2032-2012)
= 15,179-(478*9)
= 10,786

Assuming exponential decline
N_{2012} = 15,179
Decline over 20 years = 64%
Remaining = 37% = 0.37
Decline/year = $0.37^{(1/20)}$ = 0.952
N_{2021} = N_{2012}*$0.952^{(2021-2012)}$
= N_{2012}*0.952^{9}
= 15,179*0.639
= 9,704

A3 Population reductions (2021-2042):

The Committee has determined that there are insufficient data to appropriately address Criterion A3 for the koala. As above, the primary data source from which to address both population size and trend is the paper by Adams-Hosking et al. (2016). As that paper addresses the period only until three generations into the future from 2012, extending the period until 2042 would require inappropriately extrapolating by approximately a decade.

A4 Population reductions (2012-2032):

Table 4 shows a decline rate of 45 percent over the relevant three generation moving window from 2012 to 2032 (without including effects of the 2019/20 bushfires). That this is a lower overall rate than the period 2000-2021 may seem counterintuitive. This is explained by the fact that several of the highest rates of decline within bioregions occur in those bioregions with the largest population size. In earlier years, those populations constitute a higher proportion of the overall population than in subsequent years and lead to a higher overall rate of decline because they decline faster than the overall population. Consequently, as they diminish in size, they contribute less to the overall population decline and this rate itself decreases.

When the 2019/20 bushfires are factored into the declines for relevant bioregions the result approaches or exceeds the Endangered threshold, but it is difficult to quantify this because of the different data structures used in the relevant studies, particularly the absence of partitioning by bioregion by the Threatened Species Recovery Hub analysis (Legge et al. 2021) .

The Committee must also judge the likelihood of an additional event in the next decade sufficient to increase ongoing decline to $\geq$50 percent. In this context, it is notable that Australia has experienced two severe droughts in the last 20 years (Millennium Drought, Big Dry), several large scale fire events (e.g. 2009 Victorian fires, 2019/20 bushfires) and that climate models suggest both phenomena will become both more common and more severe (Di Virgilio et al. 2019; Abram et al. 2021). Consequently, given that the koala is demonstrably close to the lower threshold of Endangered and that ongoing trends suggest further events likely to be sufficient to worsen the decline, the Committee considers that the koala is eligible for listing as Endangered under subcriterion 1A4c.

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 subpopulations; (v) number of mature individuals			
(c)	Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations; (iv) number of mature individuals			

Criterion 2 evidence

Eligible under Criterion 2

Not eligible

The extent of occurrence (EOO) is estimated at 1,665,850 km^2 and the area of occupancy (AOO) is estimated at 19,428 km^2. 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).

The data presented above demonstrate the subspecies is not eligible for listing under this criterion as the EOO is > 20,000 km^2 and the AOO is > 2,000 km^2.

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 subpopulation	**≤ 50**	**≤ 250**	**≤ 1,000**
	(ii) % of mature individuals in one subpopulation =	**90 – 100%**	**95 – 100%**	**100%**
(b) Extreme fluctuations in the number of mature individuals				

Criterion 3 evidence

Eligible under Criterion 3

Not eligible

The estimated population size is > 10,000 mature individuals. The data presented above demonstrates that the koala is not eligible for listing under this criterion.

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

Eligible under Criterion 4

Not eligible

The data presented above demonstrates that the koala is not eligible for listing under this criterion. The number of individuals is > 1,000 and the AOO is > 20 km^2, and there are > 5 locations.

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

Eligible under Criterion 5 for listing as Insufficient data

Insufficient data to determine eligibility

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

Adequacy of survey

The survey and modelling 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 30 business days between 18 May 2021 and 30 July 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 transferring *Phascolarctos cinereus* from the Vulnerable category to the Endangered category.

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

Attachment B: Experts consulted with during the preparation of the Conservation Advice

Note: A National koala monitoring workshop was held February 1-2, 2021. Koala experts provided direct input to the Conservation Advice during this workshop. The workshop participants are included in the list of experts consulted with alongside other experts who provided additional advice.

Name	**Organisation/Affiliation**
Adam Leavesley	ACT Parks and Conservation
Adam Roff	NSW Department of Planning, Industry and Environment
Allen Mcllwee	NSW Department of Planning, Industry and Environment
Andrew Hoskins	CSIRO
Anthony Contarino	QLD Department of Environment and Science
Ben Moore	Western Sydney University
Bill Ellis	University of Queensland
Billie Roberts	NSW Department of Planning, Industry and Environment
Brad Law	NSW Department of Primary Industries
Bronte Kulp	QLD Department of Environment and Science
Carsten Kuelheim	Michigan Technological University, USA (formerly at: Australian National university)
Cassie Thompson	NSW Natural Resources Commission
Catherine George	QLD Department of Environment and Science
Chris Meakin	Commonwealth Department of Agriculture, Water and the Environment
Sue Fyfe	Commonwealth Department of Agriculture, Water and the Environment
Claire Runge	University of Queensland
Christine Hosking	The University of Queensland
Cristina Vicente	SA Department for Environment and Water

Damian Higgins	University of Sydney
Dan Lunney	NSW Department of Planning, Industry and Environment
Danielle Stocks	NSW Department of Planning, Industry and Environment
David Ramsey	Vic Department of Environment, Land, Water and Planning
David Westcott	CSIRO
Debbie Saunders	Wildlife Drones
Desley Whisson	Deakin University
Emma Hickingbotham	Vic Department of Environment, Land, Water and Planning
Enhua Lee	NSW Department of Planning, Industry and Environment
Grant Hamilton	Queensland University of Technology
Harriet Preece	QLD Department of Environment and Science
Helen Murphy	CSIRO
Helene Marsh	Threatened Species Scientific Committee
Ian Sandford	QLD Department of Environment and Science
Ivan Lawler	Commonwealth Department of Agriculture, Water and the Environment
Jane DeGabriel	NSW Department of Planning, Industry and Environment
Jacob Tangey	QLD Department of Environment and Science
Jennie Mallela	Commonwealth Department of Agriculture, Water and the Environment
Jim Adams	National Landcare Network
John Turbill	NSW Department of Planning, Industry and Environment
Jonathan Rhodes	University of Queensland
Julie Anorov	Commonwealth Department of Agriculture, Water and the Environment

Kaitlyn Close	QLD Department of Environment and Science
Kara Youngentob	Australian National University
Karen Ford	Australian National University
Karl Hillyard	SA Department for Environment and Water
Kath Handasyde	University of Melbourne
Katherine Belov	University of Sydney
Kellie Leigh	Science for Wildlife
Kyle Debets	QLD Department of Environment and Science
Kylie Madden	NSW Department of Planning, Industry and Environment
Lachlan Wilmott	NSW Department of Planning, Industry and Environment
Laine Edwards	Commonwealth Department of Agriculture, Water and the Environment
Laura Griffiths	Commonwealth Department of Agriculture, Water and the Environment
Lauren Smith	Commonwealth Department of Agriculture, Water and the Environment
Lily Sekuljica	NSW Department of Planning, Industry and Environment
Linda Neaves	Australian National University
Lynne McCarthy	Commonwealth Department of Agriculture, Water and the Environment
Manda Page	QLD Department of Environment and Science
Mark Eldridge	Australian Museum
Mathew Crowther	University of Sydney
Michelle Hutchins	Commonwealth Department of Agriculture, Water and the Environment
Mike Roache	NSW Department of Planning, Industry and Environment
Nerilie Abram	Australian National University

Nicholas Connor	NSW Department of Planning, Industry and Environment
Nicole Gallahar	NSW Department of Planning, Industry and Environment
Olivia Woosnam	OWAD Environment
Peter Latch	Commonwealth Department of Agriculture, Water and the Environment
Peter Menkhorst	Vic Department of Environment, Land, Water and Planning
Renae Hockey	NSW Department of Planning, Industry and Environment
Renee Brawata	ACT Government, Environment, Planning and Sustainable Development
Richard Davies	NSW Department of Planning, Industry and Environment
Rod Pietsch	NSW Department of Planning, Industry and Environment
Romane Cristescu	University of Southern Queensland
Rowan Ewing	National Landcare Australia
Ryan Witt	University of Newcastle
Sarah Bloustein	Commonwealth Department of Agriculture, Water and the Environment
Sarah Brown	Commonwealth Department of Agriculture, Water and the Environment
Sarah Legge	NESP Threatened Species Recovery Hub
Sarah Sargent	QLD Department of Environment and Science
Shane Norrish	National Landcare Australia
Steven Howell	QLD Department of Environment and Science
Tanya Pritchard	WWF
Vural Yazgin	Vic Department of Environment, Land, Water and Planning
Warrick McGrath	Vic Department of Environment, Land, Water and Planning
Zoe Kemp	QLD Department of Environment and Science

Attachment C: Additional Sources of Information Provided during the Public Consultation

Note: Additional sources of information provided during the public consultation process, that are not referred to in the Conservation Advice, are detailed here. Each has been considered with respect to finalising the Committee's recommendation and whether it materially affected the outcome or the recommended conservation actions. The inclusion of a source here does not necessarily indicate that the Committee agrees with its conclusion(s).

Biolink (2017). Koala Habitat & Population Assessment: Lismore Local Government Area (part). Final report to Lismore City Council. Biolink Ecological Consultants, Uki NSW.

Biolink (2019) The Kiwarrak and Khappinghat ARKS: Aspects of the distribution and abundance of koalas. Final report to mid Coast Council. Biolink Ecological Consultants, UKI NSW.

Biolink (2020) A review of the conservation status of the NSW populations of the koala (Phascolarctos cinereus) leading up to and including part of the 2019/20 fire event.

Biolink (2020) A review of the conservation status of the QLD population of the koala (Phascolarctos cinereus) leading up to and including the 2019 fire events. Available at: file:///C:/Users/A27456/Downloads/A%20Review%20of%20the%20Conservation%20Status%20of%20QLD%20Koalas.pdf

Biolink 2020 Burleigh Ridge koala survey data analysis & Population Viability Analysis. Final Report to City of Gold Coast May 2020. Revised August 2020

Biolink Pty Ltd (2020). Quantifying the impacts of bushfire on populations of wild koalas (Phascolarctos cinereus): Insights from the 2019/20 fire season. Final report to WWF-Australia. Available at: https://www.wwf.org.au/ArticleDocuments/353/Quantifying%20the%20impacts%20of%20bushfire%20on%20koalas,%20insights%20from%20the%202019%202020%20fire%20season%20Report_1Septv3.pdf.aspx?OverrideExpiry=Y

Biolink Ecological consultants (2017) East Coomera Koala Population Study. Prepared for the City of Gold Coast. Available at: https://www.goldcoast.qld.gov.au/files/sharedassets/public/pdfs/brochures-amp-factsheets/rti18-19005.pdf.

Biolink Ecological consultants (2017) Parkwood-Coombabah Koala Population Study 2017. Prepared for the City of Gold Coast. Available at: https://www.goldcoast.qld.gov.au/files/sharedassets/public/pdfs/brochures-amp-factsheets/rti18-19198.pdf.

Biolink Ecological consultants (2019) City-wide Koala Monitoring: Habitat mapping and Monitoring Program. Final Report to City of Gold Coast Council. Available at: https://www.goldcoast.qld.gov.au/files/sharedassets/public/pdfs/minutes-amp-agendas/economy-20191106-koala.pdf

Callaghan, J., McAlpine, C., Thompson, J., Mitchell, D., Bowen, M., de Jong, C., Rhodes, J., Sternberg, R. and Scott, A. 2011. Ranking and mapping koala habitat quality for conservation planning on the basis of indirect evidence of tree-species use: a case study of Noosa Shire, south-eastern Queensland. Wildlife Research 38: 89-102.

Crowther MS, McAlpine CA, Lunney D, Shannon I and Bryant JV (2009) Using broad-scale, community survey data to compare species conservation strategies across regions: A case study of the Koala in a set of adjacent 'catchments'. Ecological Management and Restoration 10 (S1): 88-96.

City of Gold Coast (2021). Flora and Fauna. Available at: http://www.goldcoastflorafauna.com.au/.

Coast Adapt (2018). Coast Adapt datasets. Available at: https://coastadapt.com.au/tools/coastadapt-datasets#future-datasets.

DPIE (2019) Koala Habitat Information Base technical Guide (https://www.environment.nsw.gov.au/-/media/OEH/Corporate-Site/Documents/Animals-and-plants/Threatened-species/koala-habitat-information-base-technical-guide-190534.pdf

DPIE (2020) Post-Fire Koala Surveys. A Saving our Species Project. Northeast NSW. Department of Planning, Industry and Environment, Coffs Harbour.

Department of Planning, Industry and Environment (2021). NSW Far South Coast post-fire koala survey 2020. Saving our Species, (https://vvww.environmentnsw.gov.au/-/media/OEH/Corporate-Site/Documents/animals-and-plants/Threatened-species/nsw-far-south-coast-post-fire-koala-survey-2020-2003 '12. pdf).

Information on planning, management and recovery actions for the Koala are in the SEQ Koala Conservation Strategy 2020-2025

IUCN (2009). Koalas and Climate Change: Hungry for CO2 cuts. IUCN Species Survival Commission, Gland, Switzerland.

Jurskis V (2017) Ecological history of the koala and implications for management. Wildlife Research 44(6–7), 471-483. https://doi.org/10.1071/WR17032

Jurskis V (2020) The Great Koala Scam, Connor Court.

Lane, A., Wallis, K., and Phillips, S. (2020). A review of the conservation status of New South Wales populations of the Koala (Phascolarctos cinereus) leading up to and including part of the 2019/20 fire event. Report to International Fund for Animal Welfare (IFAW). Biolink Ecological Consultants, Uki NSW. (including amendments post-bushfires).

Law B, Gonsalves L, Bilney R, Peterie, J, Pietsch R, Roe P and Truskinger A (2019) Using Passive Acoustic Recording and Automated Call Identification to Survey Koalas in the Southern Forests of New South Wales.

Law B., Gonsalves L, Burgar J., Brassi, T., Kerr I., O'Loughlin C., Eichinski P., Roe P. (2021) Regulated timber harvesting does not reduce koala density in north-east forests of New South Wales. NSW Primary Industries Unpubl. Report to Natural Resources Commission.

Law B., Gonsalves L, Burgar J., Brassil T., Kerr I., Wilmott L., Madden K., Smith M., Mella V., Crowther M., Krockenberger M., Rus, Pietsch R., Truskinger A., Eichinski P., & Roe P. (2021) Validation of Spatial Count Models to Estimate Koala Phascolarctos cinereus Density from Acoustic Arrays. NSW Primary Industries, Unpubl. Report to NSW Environmental Trust.

Law B, Caccamo G, Roe P, Truskinger A, Brassil T, Gonsalves L, Mcconville A, Stanton M (2017) Development and field validation of a regional, management-scale habitat model: A koala Phascolarctos cinereus case study. Ecology and Evolution 7:7475-7489. https://doi.org/10.1002/ece3.3300

Law BS, Brassil T, Gonsalves L, Roe P, Truskinge1- A, et al. (2018) Passive acoustics and sound recognition provide new insights on status and resilience of an iconic endangered marsupial (koala Phascolarctos cinereus) to timber harvesting. PLOS ONE 13(1 e0205075. https://doi.org/10.1371/journal.pone.0205075

Lee E, Madden K. and Wilmott L. Application of the koala spotlighting survey method in the Campbelltown area for estimates of koala densities and total population size. In The Abstracts, Australian Mammal Society Conference 2019. https://australianmammals.org.au/conferences/conference-2019.

Lunney et al., (1990), editors. Koala Summit: managing koalas in New South Wales. Proceedings of the koala Summit held at the University of Sydney 7-8 November 1988. NSW NPWS.

Lunney, D., Phillips, S., Callaghan, J. and Coburn, D. 1998. Determining the distribution of koala habitat across a shire as a basis for conservation: a case study from Port Stephens, New South Wales. Pacific Conservation Biology 4: 186-96.

Lunney, D. Sonawane, I., Wheeler, R. Tasker, E., Ellis, M., Predavec, M., Fleming, M. (2020). An ecological reading of the history of the koala population of Warrumbungle National Park. Proceedings of the Linnean Society of New South Wales 141 Supplement, S131-S154.

Lunney, D., Predavec, M., Sonawane, I., Kavanagh, R., Barrott-Brown, G., Phillips, S., Callaghan, J., Mitchell, D., Parnaby, H., Paull, D.C. (2017). The remaining koalas (Phascolarctos cinereus) of the Pilliga forests, north-west New South Wales: refugial persistence or a population on the road to extinction? Pacific Conservation Biology 23, 277-294. Doi: 10.1071/PC17008

Matusick and Fontaine (unknown) Causes of large-scale eucalyptus tree dieback and mortality: research priorities. A report for the NSW Natural Resources Commission Prepared By: Dr George Matusick and Dr Joe Fontaine

McAlpine, C., Bowen, M., Callaghan, J., Lunney, D., Rhodes, J., Mitchell, D., Pullar, D., Possingham, H.P. 2006. Testing alternative models for the conservation of koalas in fragmented rural-urban landscapes. Austral Ecology 31: 529-544.

McAlpine, C.A., Rhodes, J.R., Callaghan, J., Bowen, M., Lunney, D., Mitchell, D., Pullar, D., Possingham, H.P. 2006. The importance of forest area and configuration relative to local habitat factors for conserving forest mammals: A case study of koalas in Queensland, Australia. Biological Conservation 132: 153-165.

McAlpine, C.A., Rhodes, J.R., Peterson, A., Possingham, H.P., Callaghan, J., Curran, T., Mitchell, D., and Lunney, D. 2007. Planning Guidelines for Koala Conservation and Recovery-a guide to best planning practice. Australian Koala Foundation and the University of Queensland, Brisbane, Australia. http://espace.library.uq.edu.au/view/UQ:124088.

McAlpine, C.A., Rhodes, J.R., Bowen, M., Lunney, D., Callaghan, J., Mitchell, D., and Possingham, H.P. 2008. Can multiscale models of species' distribution be generalised from region to region? A case study of the koala. Journal of Applied Ecology 45(2): 558-567.

Moore BD, Lawler IR, R. WI, Beale C, Foley WJ (2010) Palatability mapping: a koala's eye view of spatial variation in habitat quality. Ecology 91: 3165-3176.

NSW Office of Environment and Heritage (2014). New England North West Climate Change. Available at: https://climatechange.environment.nsw.gov.au/Climate-projections-for-NSW/Climate-projections-for-your-region/New-England-North-West-Climate-Change-Downloads

Paull, et al. (2019). Koala habitat conservation plan. Report prepared for WWF-Australia, Sydney.

Phillips, S. and Callaghan, J. 2000. Tree species preferences of koalas (Phascolarctos cinereus) in the Campbelltown area south of Sydney, New South Wales. Wildlife Research 27: 502-516.

Phillips, S., Callaghan, J. and Thompson, V. 2000. The tree species preferences of koalas (Phascolarctos cinereus) inhabiting forest and woodland communities on Quaternary deposits in the Port Stephens area, New South Wales. Wildlife Research 27: 1-10.

Phillips, S. and Callaghan, J. 2011. The Spot Assessment Technique: a tool for determining localised levels of habitat use by Koalas. Australian Zoologist. 35(3): 774-80.

Phillips et al 2021 Quantifying the impacts of bushfire on populations of wild koalas (Phascolarctos cinereus): insights from the 2019/20 fire season.

Phillips et al., (2021). Quantifying the impacts of bushfire on populations of wild koalas (Phascolarctos cinereus): Insights from the 2019/20 fire season. Ecological Management & Restoration, 22, 80-88. Doi: 10.1111/emr.12458 [note this reference may relate to text at the bottom of the table on page 24]

Phillips, P., Wallis, K., Lane, A. (2021) Quantifying the impacts of bushfire on populations of wild koalas (Phascolarctos cinereus): Insights from the 2019/20 fire season. Ecological Management & Restoration 22(1), 80-88.

Predavec, M., Lunney, D., Shannon, I., Lemon, J., Sonawane, I., Crowther, M. (2017). Using repeat citizen science surveys of koalas to assess their population trend in the north-west of New South Wales: scale matters. Australian Mammalogy 40, 47-57. Doi: 10.1071/AM16059.

Queensland Government (2020) Spatial modelling for Koalas in south east Queensland. Available at: https://environment.des.qld.gov.au/__data/assets/pdf_file/0020/211772/spatial-modelling-koalas-seq-vers1-1.pdf

Reed, P and Lunney, D, 1990. Habitat loss: the key problem for the long-term survival of koalas in New South Wales, pp9-35 in Lunney et al. 1990 (ed) Koala Summit: managing koalas in New

South Wales. Proceedings of the Koala Summit held at the University of Sydney 7-8 November 1988. NSW NPWS.

Rhodes, J. R., Beyer, H. L., Preece, H.J. and McAlpine, C.A. 2015. South East Queensland Koala Population Modelling Study. UniQuest, Brisbane, Australia.

Rhodes, J.R., Callaghan, J., McAlpine, C., de Jong, C., Bowen, M., Mitchell, D., Lunney, D., and Possingham, H.P. 2008. Regional variation in habitat-occupancy thresholds: a warning for conservation planning. Journal of Applied Ecology 45(2): 549-557.

Rhodes, J. R., Wiegand, T., McAlpine, C.A., Callaghan, J., Lunney, D., Bowen, M., and Possingham, H.P. 2006. Modelling species distributions to improve conservation in semiurban landscapes: koala case study. Conservation Biology 20: 449-459.

Taylor, M, and Blanch, S (2019). Koalas face extinction in Eastern Australia, a deforestation hotspot. Briefing prepared by WWF-Australia, Sydney, 6 pp.

Taylor, M, 2020. Destruction of koala habitat increased after listing as vulnerable in 2012. Report by WWF-Australia, Sydney, 13 pp.

Santika, T., McAlpine, C. A., Lunney, D., Wilson, K. A., and Rhodes, J. R. 2014. Modelling species distributional shifts across broad spatial extents by linking dynamic occupancy models with public-based surveys. Diversity and Distributions 20:786-796.

Santika T. C.A. McAlpine, D. Lunney K.A. Wilson, and J.R. Rhodes. 2015. Assessing spatio-temporal priorities for species' recovery in broad-scale dynamic landscapes. Journal of Applied Ecology 52: 832-840. Doi'. 10.1111/1365-2664.12441

Wallis, K., Lane, A. and Phillips, S. (2020). A review of the conservation status of Queensland populations of the Koala (Phascolarctos cinereus) arising from events leading up to and including the 2019 fire events. Report commissioned by the World Wide Fund for Nature (WWF) Australia, in collaboration with Humane Society International (HSI) and International Fund for Animal Welfare (IFAW). Biolink Ecological Consultants, Uki NSW.

Wilmott L (2020). Talk: Koala home range size and chlamydia disease expression correlated with soil fertility. Ecological Society of Australia conference 2020. https://www.esa2020.org.au/full-scientific-program/.

Witt RR, Beranek CT, Howell LG, Ryan SA, Clulow J, Jordan NR, et al. (2020) Real time drone derived thermal imagery outperforms traditional survey methods for an arboreal forest mammal. PLoS ONE 15(11): e0242204. https://doi.org/10.1371/journal.pone.024220.

Other sources

Koala likelihood map - https://researchdata.edu.au/nsw-koala-likelihood-august-2019/1426089

Koala research in NSW forests - https://www.dpi.nsw.gov.au/forestry/science/koala-research

Koala populations and habitat in NSW - https://www.parliament.nsw.gov.au/lcdocs/inquiries/2536/Koala%20populations%20and%20habitat%20in%20New%20South%20Wales%20-%20Report%203.pdf

https://www.goldcoast.qld.gov.au/files/sharedassets/public/pdfs/minutes-amp-agendas/planning-20210318-adoptedminutes_part3.pdf

https://www.environment.nsw.gov.au/topics/animals-and-plants/threatened-species/programs-legislation-and-framework/nsw-koala-strategy/building-knowledge-on-koala-habitat

https://www.parliament.nsw.gov.aullcdocslinquiries/2.536/Koala%20populations%20and%20habitat%20in%20New%20South%20Wa!es%20-%20Report%203.pdf

Cataloguing data

This publication (and any material sourced from it) should be attributed as: Department of Agriculture, Water and the Environment 2022, Conservation advice for *Phascolarctos cinereus* (Koala), Canberra.

This publication is available at the SPRAT profile for *Phascolarctos cinereus* (Koala).

Department of Agriculture, Water and the Environment
GPO Box 858, Canberra ACT 2601
Telephone 1800 900 090
Web awe.gov.au

Version history table

Document type	Title	Date
Conservation Advice	Department of Sustainability, Environment, Water, Population and Communities (2012). Approved Conservation Advice for Phascolarctos cinereus (combined populations in Queensland, New South Wales and the Australian Capital Territory). Canberra: Department of Sustainability, Environment, Water, Population and Communities.	02 05 2012
Listing Advice	Threatened Species Scientific Committee (TSSC) (2012). Listing advice for Phascolarctos cinereus (Koala)	02 05 2012

The Minister's delegate approved this conservation advice on 01/10/2015

Conservation Advice

Tyto novaehollandiae kimberli

masked owl (northern)

Conservation Status

*Tyto novaehollandiae kimberli (*masked owl (northern)) is listed as Vulnerable under the *Environment Protection and Biodiversity Conservation Act 1999* (Cwlth) (EPBC Act). The species is eligible for listing as Vulnerable as, prior to the commencement of the EPBC Act, it was listed as Vulnerable under Schedule 1 of the *Endangered Species Protection Act 1992* (Cwlth).

The Action Plan for Australian Birds 2010 (Garnett et al., 2011) list the masked owl (northern) as Vulnerable. The main factors that the Action Plan for Australian Birds 2010 identifies as making the subspecies as eligible for listing in the Vulnerable category are a limited number of mature individuals (approximately 3000), a suspected continuing decline in population size and a geographic distribution that may be precarious for the survival of the species (Garnett et al., 2011).

Description

The masked owl (northern) is a large owl with a prominent heart-shaped facial disc and plumage that is highly patterned by speckling and is generally darker on the back and paler below (Woinarski, 2004). The northern subspecies and the Tiwi Islands subspecies (*T. n. melvillensis)* of masked owl are smaller than other Australian subspecies (Woinarski, 2004), including the nominate subspecies (*T. n. novaehollandiae*) which can reach lengths of up to 41 cm and 50 cm with wings spans of up to 110 cm and 128 cm (male and female sizes respectively) (Higgins & Peter, 2002). Compared to other species of *Tyto* owls in northern Australia, such as the barn owl (*T. alba)*, masked owls have conspicuously well feathered legs and large, strong claws and feet (Higgins & Peter, 2002).

Distribution

The distribution of the masked owl (northern) is very poorly known (Woinarski 2004). Three subpopulations have been suggested: Kimberley, Northern Territory and Cape York (Garnett et al., 2011).

The few records that are available from the Kimberley region of Western Australia show the masked owl (northern) to occur from Yampi Sound north-east to Cambridge Gulf, including Windjana Gorge and Augustus Island (Barrett et al., 2003; Johnstone & Storr, 1998; Mees, 1964). There are also historical records from near Broome (Crossman, 1910).

In the Top End of the Northern Territory, the species occurs from the Cobourg Peninsula down to Katherine and Jasper Gorge (Victoria River area), and to the east at McArthur River. There are also records from Dead Dog Waterhole (Barkly Tableland) and the Tanami Desert (Barrett et al., 2003; Blakers et al., 1984; Goodfellow, 2001; Higgins, 1999; Mees, 1964).

In Queensland, there are historical records from the Normanton region, and from Pascoe, Archer, Chester and Watson Rivers on Cape York Peninsula (Higgins, 1999; Mees, 1964; Storr, 1984). The owl occurs along the southern rim of the Gulf of Carpentaria, Cape York Peninsula and south to Atherton Tablelands and the Einasleigh-Burdekin divide (Garnett et al., 2011). There is some confusion about where the Queensland southern limit of the subspecies is, with authorities suggesting Mackay (Mees, 1964) or Coomooboolaroo Station (west of Rockhampton) (Woinarski, 2004).

Threats

The reason for the decline and low density of masked owls in northern Australia is unclear. The subspecies has undoubtedly been affected by broad-scale changes to the environment of northern Australia caused by altered fire regimes, grazing by livestock and feral animals, and the invasion of native woodlands by exotic plants, particularly introduced pasture grasses (Woinarski, 2004). However, the most likely cause of declines is a shortage of food, as small and medium-sized native mammals are becoming increasingly uncommon across much of northern Australia (Pardon et al., 2003; Sattler & Creighton, 2002; Winter & Allison, 1980; Woinarski et al., 2001; Woinarski et al., 2010).

The current regime of more intense, frequent and extensive fires may also reduce the availability of the large trees and hollows (Williams et al., 1999) required for nesting. One study in tall eucalypt forests and woodlands near Darwin (Pittman, 2003) found that the populations of common brushtail possums (*Trichosurus vulpecula)* and black-footed tree-rats (*Mesembriomys gouldii)* were nearing a carrying capacity imposed by hollow availability, and possums were found to monopolise hollows in woodland fragments at the expense of other species.

Other potential threats include competition with other large owls (Schodde & Mason, 1980) and the increasing spread and pace of development in the Darwin and Daly River areas of the Northern Territory, which could be reducing the extent of suitable habitat for the subspecies (Woinarski, 2004).

Conservation Actions

Conservation and management actions

- Implement an appropriate fire management regime for preventing the loss of large, hollow-bearing trees, and which promotes the density of prey (native mammals).
- Reduce the impacts from feral animals and weeds at a landscape scale.

Survey and monitoring priorities

- Assess the subspecies' population size and distribution.
- Design and implement a monitoring program to assess population trends at key sites.

Information and research priorities

- Identify the habitat requirements of the subspecies.
- Assess population trends in response to fire management and weed and feral species control programs.
- Identify the causes for the decline in the masked owl's main prey species.
- Examine impacts of fragmentation on the subspecies and use the resulting knowledge to develop guidelines for habitat protection and corridor configuration in landscapes subject to increasingly intensive development.

References cited in the advice

Barrett G, Silcocks A, Barry S, Cunningham R and Poulter R (2003). The New Atlas of Australian Birds. Birds Australia: Melbourne, Victoria.

Blakers M, Davies SJFF and Reilly PN (1984). The Atlas of Australian Birds. Melbourne University Press: Melbourne, Victoria.

Crossman AF (1910). Birds seen in and around Broome, north-western Australia. Emu. 10: 111-113.

Garnett ST and Crowley GM (2000). The Action Plan for Australian Birds 2000. Environment Australia and Birds Australia: Canberra, ACT.

Garnett ST, Szabo J and Dutson G (2011). *The Action Plan for Australian Birds 2010*. CSIRO Publishing: Canberra, ACT.

Goodfellow DL (2001). Birds of Australia's Top End. Reed New Holland: Sydney.

Higgins PJ (Ed.) (1999) Handbook of Australian, New Zealand and Antarctic Birds. Volume 4: Parrots to Dollarbirds. Oxford University Press: Melbourne, Victoria.

Johnstone RE and Storr GM (1998). Handbook of Western Australian Birds. Vol. 1. Non-passerines (Emu to Dollarbird). Western Australian Museum: Perth, WA.

Mees GF (1964). A revision of the Australian Owls (Striigdae and Tytonidae). Zoologische Verhandelingen. 65:1--62.

Nielsen L (1996). Birds of Queensland's Wet Tropics and Great Barrier Reef, Australia. Gerard Industries, Bowden.

Pardon LG, Brook BW, Griffiths AD and Braithwaite RW (2003). Determinants of survival for the Northern Brown Bandicoot under a landscape-scale fire experiment. Journal of Animal Ecology. 72:106-115.

Pittmann GW (2003). Occurrence and Use of Tree Hollows by Mammals in Fragmented and Continuous Savanna Woodlands in Northern Australia. Hons. Thesis. BSc (Honours) thesis, Northern Territory University.

Sattler P and Creighton C (2002). Australian Terrestrial Biodiversity Audit. National Land and Water Resources Audit: Canberra, ACT.

Schodde R and Mason IJ (1980). Nocturnal Birds of Australia. Lansdowne: Melbourne, Victoria.

Storr GM (1984). Revised list of Queensland birds. Records of the Western Australian Museum Supplement. 19:1-189.

Williams RJ, Cook GD, Gill AM and Moore PHR (1999). Fire regimes, fire intensity and tree survival in a tropical savanna in northern Australia. Australian Journal of Ecology. 24:50-59.

Winter JW and Allison FR (1980). The native mammals of Cape York Peninsula – changes in status since the 1948 Archibold Expedition. In Contemporary Cape York Peninsula. (Eds NC Stevens and A Bailey) pp. 31–44. Royal Society of Queensland, St Lucia, Queensland.

Woinarski JCZ (2004). National multi-species recovery plan for the Partridge Pigeon [eastern subspecies] *Geophaps smithii smithii*, Crested Shrike-tit [northern (sub)species] *Falcunculus (frontatus) whitei*, Masked Owl [north Australian mainland subspecies] *Tyto novaehollandiae kimberli*; and Masked Owl [Tiwi Islands subspecies] *Tyto novaehollandiae melvillensis*, 2004– 2009. Northern Territory Department of Infrastructure Planning and Environment: Darwin, NT.

Young J and De Lai L (1997). Population declines of predatory birds coincident with the introduction of Klerat rodenticide in north Queensland. Australian Bird Watcher. 17:160-167.

The Minister approved this conservation advice and included this species in the Critically Endangered category, effective from 04/07/2019.

Conservation Advice

Cophixalus mcdonaldi

(Mount Elliot Nursery Frog)

Taxonomy

Conventionally accepted as *Cophixalus mcdonaldi* (Zweifel 1985).

Summary of assessment

Conservation status

Critically Endangered: Criterion 2 B1 (a),(b)(i,ii,iii,v)

The highest category for which *Cophixalus mcdonaldi* is eligible to be listed is Critically Endangered.

Cophixalus mcdonaldi has been found to be eligible for listing under the following categories:
Criterion 2: B1 (a),(b)(i,ii,iii,v): Critically Endangered

Cophixalus mcdonaldi has been found to be eligible for listing under the Critically Endangered category.

Species can be listed as threatened under state and territory legislation. For information on the listing status of this species under relevant state or territory legislation, see http://www.environment.gov.au/cgi-bin/sprat/public/sprat.pl

Reason for conservation assessment by the Threatened Species Scientific Committee

This advice follows assessment of new information provided to the Committee to list *Cophixalus mcdonaldi.*

Public consultation

Notice of the proposed amendment and a consultation document was made available for public comment for 30 business days between 7 September 2018 and 22 October 2018. Any comments received that were relevant to the survival of the species were considered by the Committee as part of the assessment process.

Species Information

Description

The Mount Elliot Nursery Frog is smooth and pale to dark brown above with scattered darker markings, which typically include a dark streak above each arm, an obscure interorbital bar, dark canthal and temporal streaks and dark facial markings. It is smooth and pale underneath with dark stippling and mottling and the discs of the fingers and toes are well developed (Cogger 2014). Males are up to 23 mm snout-to-vent length (SVL) in size and females up to 26 mm SVL (Zweifel 1985). The male call is a short trill that differs from all other Australian *Cophixalus* species (Hoskin 2004).

The eggs of microhylids are relatively large and are laid in very moist soil. The tadpole develops inside the egg and when it has completed metamorphosis it hatches from the egg as a fully formed froglet (Zweifel 1985). One gravid female Mount Elliot Nursery Frog was found to contain 17 eggs (Anstis 2017).

Distribution

The Mount Elliot Nursery Frog is found only on Mount Elliot, south-east of Townsville (Zweifel 1985; Hoskin 2004; Williams 2007). Mount Elliot is the highest mountain in the region, located in the Bowling Green National Park and is relatively undisturbed. The Mount Elliot Nursery Frog has only been recorded in areas from 900 m and above (Hoskin 2004). The population occurs just outside the Wet Tropics biogeographic region of northern Queensland.

Relevant Biology/Ecology

Very little is known of the specific biology of the Mount Elliot Nursery Frog. The species is most closely related to *C. neglectus* (Williams 2007) and like *C. neglectus* and *C. concinnus*, it is a high altitude rainforest-restricted species (Shoo & Williams 2004). Frogs have been found during the day sheltered in fallen palm fronds and beneath rocks, with the males emerging to call in the late afternoon (Hoskin 2004). Males concentrate around rocky creek margins and call from the ground level, or close to it (Hoskin 2004). Individuals have also been found in rotted tree stumps, under flat rocks and in rock cracks, with a clutch of up to eight eggs located inside a small hole (2 cm in diameter) in a solid rock face (Williams et al. 1993).

The microhylids of the Australian Wet Tropics differ from most other frog species in that they are terrestrial breeders and do not need surface water to breed. They require high levels of soil and litter moisture to prevent dessication of the eggs during development (Williams 2007). One parent (usually the male) will generally attend to the eggs until hatching occurs (Felton et al. 2006; Hoskin 2004; Williams 2007). The embryo develops directly in the egg and then hatches out as a tiny froglet. The eggs are large relative to other frog species and clutch sizes are small (Hoskin 2004).

The generation length of the Mount Elliot Nursery Frog is unknown, but is estimated to be 10 years, based on the known ages of breeding males being between 4-14 years for *Cophixalus ornatus* (Ornate Nursery Frog) (Felton et al. 2006).

Threats

Threats to the Mount Elliot Nursery Frog include climate change, habitat degradation and introduced species. The table below lists the threats impacting the species in approximate order of severity of risk, based on available evidence.

Number	Threat factor	Threat status	Evidence base
1.0	Climate change		
1.1	Temperature increase, extreme weather events e.g. cyclones, droughts	Known potential	The Mount Elliot Nursery Frog is found only at high altitude on a single mountain top in northern Queensland. Distribution modelling for congeneric species suggests it could lose a substantial proportion of its available habitat due to climate change (Williams et al. 2003; Meynecke 2004; Shoo 2005; Williams & Hilbert 2006). Climate change modelling carried out by Williams and Hilbert (2006) suggests that five *Cophixalus* species would lose more than 50

			percent of their core habitat with a 1 ^{o}C increase in temperature. However an increase by 3 - 5 ^{o}C is predicted to be more likely in the next 50 years. Changes in hydrology and other effects of climate change (e.g. reduction in food supply) may also alter the susceptibility of frogs to disease, but these impacts are likely to be variable among species and sites (DoEE 2016).
2.0	Habitat loss and degradation		
2.1	Clearing, trampling, fragmentation, altered hydrology	Known potential	Feral pigs are responsible for habitat damage and potentially cause adult frog mortality (Richards et al. 1993).
3.0	Invasive species		
3.1	Yellow Crazy Ants *(Anoplolepis gracilipes)*	Known potential	Yellow crazy ants spray formic acid to subdue prey, which causes burns and irritates the skin and eyes of animals. They can have severe impacts on a range of ecological processes and lead to significant loss of biodiversity. Yellow crazy ants were detected within the World Heritage Area and Little Mulgrave National Park in 2012 and now cover up to 61 ha (WTMA 2016) within these protected areas. In December 2013 yellow crazy ants were also detected in the Kuranda area (WTMA 2016).
4.0	Disease		
4.1	Amphibian chytrid fungus	Known current	Chytridiomycosis is an infectious disease caused by the amphibian chytrid fungus (*Batrachochytrium dendrobatidis*) that affects amphibians worldwide, causing mass die-offs and some species extinctions (DoEE 2016). However, the prevalence of chytrid is extremely low in Australian microhylids (Hauselberger & Alford 2012).

How judged by the Committee in relation to the EPBC Act Criteria and Regulations

Criterion 1. Population size reduction (reduction in total numbers) Population reduction (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.	*based on any of the followin*	(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
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.		

Evidence:

Insufficient data to determine eligibility

Given that the generation length of the Mount Elliot Nursery Frog is estimated to be approximately 10 years, the appropriate time scale for this criterion is likely to be 30 years. There are no data available to evaluate the population trend over any three generation period.

The species may experience natural fluctuations in number due to seasonal and climatic variation and there is insufficient information to conclude whether or not the observed changes in population size are a result of natural fluctuations. The available data does not allow a quantitative estimate of decline, therefore the Committee considers that there is insufficient information to determine the eligibility of the species for listing in any category under this criterion.

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 km^2**	**< 5,000 km^2**	**< 20,000 km^2**
B2. Area of occupancy (AOO)	**< 10 km^2**	**< 500 km^2**	**< 2,000 km^2**
AND at least 2 of the following 3 conditions indicating distribution is precarious for survival:			
(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 subpopulations; (v) number of mature individuals			

(c)	Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations;(iv) number of mature individuals

Evidence:

Eligible under Criterion 2 B1 (a),(b)(i,ii,iii,v) for listing as Critically Endangered

The calculated extent of occurrence (EOO) is 14 km^2, and the area of occupancy (AOO) is 12 km^2 (unpublished data DoEE 2017). These figures are based on the mapping of point records from post-1997 (20 year timeframe), compiled from state and Commonwealth agencies along with museums, research institutions and non-government organisations. 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. The EOO meets the threshold for listing as Critically Endangered under subcriterion B1 and the AOO meets the threshold for listing as Endangered under subcriterion B2.

There is a single population on Mt Ellliot in northern Queensland (Zweifel 1985; Hoskin 2004) where climate change would be expected to have major impacts.

A continuing decline in area of occupancy and area, extent and/or quality of habitat, and therefore number of mature individuals, may be inferred based on climate change (Shoo 2005; Williams et al. 2003; Williams and Hilbert 2006). Species that are both geographically restricted and patchily distributed, such as *C. mcdonaldi*, are at a high risk of extinction, as local stochastic events may affect the entire population (Williams 2007).

The Committee considers that the species' extent of occurrence is very restricted, the area of occupancy is restricted and the geographic distribution is precarious for the survival of the species because it occurs at only one location and a decline in area of occupancy and area, extent and/or quality of habitat and number of mature individuals has been inferred. Therefore, the species has met the relevant elements of Criterion 2 to make it eligible for listing as Critically Endangered.

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 subpopulation	**≤ 50**	**≤ 250**	**≤ 1,000**
(a) (ii) % of mature individuals in one subpopulation =	**90 – 100%**	**95 – 100%**	**100%**
(b) Extreme fluctuations in the number of mature individuals			

Evidence:

Insufficient data to determine eligibility

There are no data available to assess population size. Therefore, there are insufficient data to demonstrate if the species is eligible for listing under Criterion 3.

Criterion 4. Number of mature individuals			
	Critically Endangered Extremely low	**Endangered Very Low**	**Vulnerable Low (Medium-term future)[1]**
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 km2 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 that demonstrate eligibility for listing under other criteria may include information relevant to D2. This information will not be considered by the Committee in making its assessment 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.

Evidence:

Insufficient data to determine eligibility

There are no data available to assess population size. Therefore, there are insufficient data to demonstrate if the species is eligible for listing under Criterion 4.

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**

Evidence:

Insufficient data to determine eligibility

Population viability analysis has not been undertaken. Therefore, there are insufficient data to demonstrate if the species is eligible for listing under Criterion 5.

Conservation Actions

Recovery Plan

A recovery plan is not recommended because the Mount Elliot Nursery Frog is located in a small area in a single jurisdiction and the Conservation Advice sufficiently outlines the priority research and conservation actions needed to support the recovery of this species.

Conservation and Management priorities

Habitat loss and disturbance

- Implement a program ensuring suitable habitat is maintained in areas currently supporting populations of the Mount Elliot Nursery Frog and investigate options for enhancing the resilience of the species' current habitat to climate change.

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

- Reduce the impacts of habitat destruction by feral pigs on existing populations by using fencing (where feasible) and reducing pig numbers.
- Control yellow crazy ants by baiting at critical stages of the ants' life cycle.

Disease

- Minimise the spread of the amphibian chytrid fungus by implementing suitable hygiene protocols (Murray 2011) to protect priority populations as described in the *Threat abatement plan for infection of amphibians with chytrid fungus resulting in chytridiomycosis* (Department of the Environment and Energy 2016).
- Provide disease identification and prevention protocols (methods of handling, diagnostic keys, etc.) to researchers and land managers for use in the field.

Stakeholder Engagement

- Collaborate with land managers bordering (outside of) the Wet Tropics World Heritage Area to protect and manage rainforest areas where the species occurs, or which contain potential habitat for the species, from threats due to disease and invasive species.
- Interested nature conservation, land management and land holder groups could be engaged in conservation management activities, such as survey and monitoring, but should be made aware of the need to follow correct field practices and hygiene protocols to mitigate the risks of trampling and disease transmission. If necessary, use workshops to aid stakeholders in developing the skills and knowledge required to manage threats to this species while undertaking these activities.
- Inform the public about the status and recovery efforts for the species, e.g. by providing information to visitors to the Wet Tropics World Heritage Area and publicising the species through the media.

Survey and Monitoring priorities

- More precisely assess the population size, distribution and ecological requirements of the Mount Elliot Nursery Frog.
- Design and implement a monitoring program for the Mount Elliot Nursery Frog.

Information and research priorities

- Improve knowledge of the reproductive biology, age structure and growth rates of the Mount Elliot Nursery Frog.

- Improve knowledge of the thermal tolerance limits of the Mount Elliot Nursery Frog and assess its possible response to future climate scenarios.
- Improve understanding of how climate change will likely impact on the Mount Elliot Nursery Frog due to altered temperatures, rainfall, environmental stressors and disease virulence.
- Improve understanding of husbandry methods for the species.
- Investigate the development of a strategic assisted colonisation (or translocation) strategy in response to the threat of climate change. The strategy should include consideration of the benefits and risks of undertaking a coordinated series of translocations of *Cophixalus* species to mountain tops further south as increased temperatures impact on their survival and reproductive success.
- Improve understanding of the impacts of feral pigs and yellow crazy ants on the Mount Elliot Nursery Frog.

Recommendations

(i) The Committee recommends that the list referred to in section 178 of the EPBC Act be amended by **including** in the list in the Critically Endangered category: *Cophixalus mcdonaldi*

(ii) The Committee recommends that there not be a recovery plan for this species.

Threatened Species Scientific Committee

26/02/2019

References cited in the advice

Anstis M (2017). *Tadpoles and Frogs of Australia* 2nd Edition. New Holland Publishers Pty Ltd, Australia.

Cogger H G (2014). Reptiles and Amphibians of Australia. 7th edn. CSIRO Publishing, Collingwood, Victoria.

Felton A R, Alford R A and Schwarzkopf L (2006). Multiple mate choice criteria and the importance of age for male mating success in the microhylid frog, *Cophixalus ornatus. Behavioural Ecology and Sciobiology* 59:786-795.

Hauselberger K F and Alford R A.(2012). Prevalence of *Batrachochytrium dendrobatidis* infection is extremely low in direct-developing Australian microhylids. *Diseases of Aquatic Organisms 100*,191-200.

Hoskin C J (2004). Australian microhylid frogs (*Cophixalus* and *Austrochaperina*): phylogeny, taxonomy, calls, distributions and breeding biology. *Australian Journal of Zoology 52*,237-269.

Hoskin C J & Higgie M (2005). Minimum calling altitudes of Cophixalus frogs on Thornton Peak, Northeastern Queensland. *Memoirs of the Queensland Museum 51*,572.

Meynecke J-O (2004). Effects of global climate change on geographic distributions of vertebrates in North Queensland. *Ecological Modelling 174*,347-357.

Richards S J, McDonald K R & Alford R A (1993). Declines in populations of Australia's endemic tropical rainforest frogs. *Pacific Conservation Biology 1*,66-77.

Shoo L (2005). Predicting and detecting the impacts of climate change on montaine fauna in Australian tropical rainforests. PhD thesis. Griffith University.

Shoo L & Williams Y (2004). Altitudinal distribution and abundance of microhylid frogs (*Cophixalus* and *Austrochaperina*) of north-eastern Australia: baseline data for detecting biological responses to future climate change. *Australian Journal of Zoology 52*,667-676.

Williams S E, Bolitho E E & Fox S (2003). Climate change in Australian tropical rainforests: an impending environmental catastrophe. *Proceedings of the Royal Society of London B 270*,1887-1892.
Williams S E & Hilbert D W (2006). Climate change as a threat to the biodiversity of tropical rainforest in Australia. In: Laurance WF, CA Peres (eds) Emerging Threats to Tropical Forests. University of Chicago Press. Chicago. pp 33-53.
Williams S, Pearson R & Burnett S (1993). Vertebrate fauna of three mountain tops in the Townsville region, north Queensland: Mount Cleveland, Mount Elliot and Mount Halifax. *Memoirs of the Queensland Museum* 33(1): 379-287. Brisbane.
Williams Y (2007). Ecological differences between rare and common species of microhylid frogs of the Wet tropics biogeographic region. Thesis. James Cook University.
Zweifel R G (1985). Australian frogs of the family Microhylidae. *Bulletin of the American Museum of Natural History* 182:267-388.

Other sources cited in the advice

Atlas of Living Australia (2016) http://spatial.ala.org.au/?q=lsid%3Aurn%3Alsid%3Abiodiversity.org.au%3Aafd.taxon%3A586367df-b602-44e4-b462-f0c3d0e75a48#

Department of the Environment and Energy (2016). Threat abatement plan for infection of amphibians with chytrid fungus resulting in chytridiomycosis, Commonwealth of Australia 2016. Available from: http://www.environment.gov.au/biodiversity/threatened/publications/tap/infection-amphibians-chytrid-fungus-resulting-chytridiomycosis-2016

Department of the Environment and Energy (2017). Area of Occupancy and Extent of Occurrence for *Cophixalus mcdonaldi*. Unpublished report, Australian Government Department of the Environment, Canberra.
Wet Tropics Management Authority (WTMA) (2016). Stamp Out Yellow Crazy Ants. Viewed 2 December 2016. Available on the internet at: http://www.wettropics.gov.au/stamp-out-yellow-crazy-ants.html.

Northern Quoll (Dasyurus hallucatus)

Advice to the Minister for the Environment and Heritage from the Threatened Species Scientific Committee (TSSC) on Amendments to the list of Threatened Species under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act)
12 April 2005

1. Scientific name, common name (where appropriate), major taxon group

Dasyurus hallucatus (Northern Quoll)

2. Description

The Northern Quoll is the smallest of the four Australian quoll species. It has reddish brown fur, with its underside cream, white spots on its back and rump with the tail unspotted, and a pointy snout. The Northern Quoll is a solitary carnivorous marsupial that makes its dens in rock crevices, tree holes or occasionally termite mounds, and is predominantly nocturnal. Northern Quolls can weigh up to 1.2kg, with the males being larger than the females. They breed only once in the middle of the year with an average litter size of seven, the pups being born in the dry season. Northern Quolls have short life spans, with males living to approximately 1 year and the oldest recorded female in the wild being three years of age. In savanna landscapes, females maintain territories of about 35 hectares, with males estimated to range over 150 hectares. Northern Quolls consume a wide range of prey including beetles, grasshoppers, spiders, scorpions and centipedes. They also eat fruit, nectar, and are known to feed on carrion and human refuse. Northern Quolls hunt a wide variety of vertebrates including the Northern Brown Bandicoot, the Common Brushtail Possum, rats, Sugar Gliders, insectivorous bats, quails, bird eggs, snakes and frogs. Frogs are eaten regularly in the wet season with a least seven species taken, including large native frogs such as the Green Tree Frog, *Litoria caerulea*, and the Giant Frog, *Cyclorana australis*. As Cane Toads have moved into the Northern Quoll's range, they have also been added to the diet of the Northern Quoll.

3. National context

Historically, the Northern Quoll was common across northern Australia occurring almost continuously from the Pilbara, Western Australia to near Brisbane, Queensland. The species' preferred habitat consists of rocky escarpment, open forest and open woodland. A 75% reduction in the Northern Quoll's range between 1900-1990 has been suggested such that, during this time, the Northern Quoll has been reduced to six major geographical centres: Drummond Range, central Queensland; wet tropics Northern Queensland; northern Cape York Peninsula; northern and western Top End, Northern Territory; north Kimberley and Pilbara, Western Australia (Braithwaite and Griffiths 1994). The Northern Quoll is absent from Bathurst and Melville Islands but present on other smaller islands.

The Northern Quoll is listed as Vulnerable under the *Territory Parks and Wildlife Conservation Act* (Northern Territory). It is not listed under threatened species legislation in the States of New South Wales, Queensland or Western Australia.

EPBC Act criteria.

TSSC judges the species to be eligible for listing as endangered under the EPBC Act. The justification against the criteria is as follows:

Criterion 1 - It has undergone, is suspected to have undergone or is likely to undergo in the immediate future a very severe, severe or substantial reduction in numbers.

The Northern Quoll has undergone a substantial reduction in numbers and it is considered that this decline is likely to continue in the immediate future. There may be a number of factors that have contributed to the current status of the Northern Quoll, including changes in vegetation structure, fire frequency, and the introduction of exotic herbivores. Key threats to Northern Quoll populations are considered to be more extensive and frequent fires in northern Australia; predation following fire; and more recently poisoning by the Cane Toad, *Bufo marinus*.

A range of recent ecological studies have suggested that the Northern Quoll is vulnerable to extensive frequent fires now characteristic of much of the species' north Australian range. In a detailed radio-tracking study undertaken before the arrival of Cane Toads, it was reported that the main cause of Northern Quoll mortality at Kapalga, Kakadu National Park, was predation in the period following extensive fire (Oakwood 2000).

Northern Quolls are known to mouth Cane Toads causing the release of poison from the Cane Toad's parotoid glands (the swellings on each shoulder behind the eardrum). The poison is then ingested with the symptoms of death including bright red lips and or gums and can also include a red roof of mouth or bright red nose and nose bleeds, red ears, bleeding from the ears, a red eye, red skin pouch, bright purple teats and faeces around the anus (Oakwood 2003).

Based on analyses of distributional records, one study considered that there had been a 75% reduction in the range of the Northern Quoll over the period 1900-1990 and that the species' distribution had contracted to six major centres: Drummond Range area, central Queensland; wet tropics of northern Qld; northern Cape York Peninsula; northern and western Top End of the Northern Territory; north Kimberley; and Pilbara, Western Australia (Braithwaite and Griffiths 1994).

There are no historical data on abundance. Information on the current status of the Northern Quoll derives from anecdotal reports from Queensland including Cape York Peninsula, recent field research in Kakadu National Park, Northern Territory, and anecdotal reports and estimates of the likely impact of colonising Cane Toads in Western Australia.

Queensland

It is known that Cane Toads colonised Cape York between the mid 1980s to the mid 1990s with subsequent crashes in Northern Quoll populations. During 1994-95, the disappearance of Northern Quolls was reported at two monitored sites in northern Cape York Peninsula within three months of the arrival of Cane Toads, with no subsequent population return observed (Burnett 1997). Northern Quolls had previously been common at both these sites.

In northern Queensland populations of Northern Quoll are known to have survived for over fifty years in areas invaded by Cane Toads. Northern Quolls are still present in localised pockets, such as Cape Cleveland/Mt Elliott and Mareeba, in which Cane Toads have occurred for many years and it has been observed that such populations occur in small, high altitude areas associated with extremely rocky habitats. Northern Quolls are also thought to co-exist with reasonably high local densities of Cane Toads in several coastal sites in north Queensland. There is limited quantitative data on the extent and density of the remaining populations or the precise factors that have led to the survival of these remnant populations following Cane Toad invasion.

While Northern Quolls are still present in a number of localised areas in Queensland in which Cane Toads have been present for many years, and other factors may have contributed to the species decline in Queensland, including loss of habitat and predation following fire, they do not appear, to date, to be recolonising their former locations. There is no evidence that a recovery is occurring in these populations, or is likely to occur, following Cane Toad invasion, nor is their any information on the viability of these remnant populations. In particular, Northern Quoll appear not to have recovered in savannah areas, west of the eastern escarpment and in Cape York Peninsula.

Northern Territory

One study reported a major decline in the Northern Quoll over the period 1986-1999 at Kapalga within Kakadu National Park, Northern Territory (Woinarski *et al*. 2001). More recent studies in Kakadu National Park have demonstrated that local extinction of Northern Quolls is occurring following Cane Toad invasion.

Across a broad region of the south of Kakadu National Park, trapping success for Northern Quolls at 77 monitored sites was compared immediately before and after arrival of Cane Toads including 33 sites that had not yet been invaded by Cane Toads which was trapped over the same period. In the Cane Toad invaded sites there was a highly significant decline in the abundance of Northern Quolls, but no decline observed in the control sites (Watson and Woinarski 2003).

Over much the same period, another study systematically trapped and radio-tracked individual Northern Quolls around the Mary River district of Kakadu National Park, reporting local extinction of Northern Quolls within twelve months of the arrival of Cane Toads, and demonstrating large scale mortality of Northern Quolls directly attributable to poisoning by Cane Toads. 31% of recorded deaths appeared to have been caused by Cane Toad poisoning, there being no evidence of disease, heavy parasite infestation, or any other obvious changes at the site that could be responsible for the rapid decline (Oakwood 2004).

Comparable results have now been reported following the more recent (December 2003) arrival of Cane Toads in the East Alligator district, in the north of Kakadu National Park, (Oakwood 2004), where Northern Quoll numbers declined from 45 individuals in the wet season of 2002-03 to four individuals in March 2004 (i.e. within 3 months of Cane Toads arriving). Extensive trapping (4000 trap nights at 56 widely dispersed sites) in sandstone uplands (i.e. prime quoll habitat) in the south of Kakadu in February 2003 (about two years after the invasion of Cane Toads) resulted in no Northern Quolls being found (Watson and Woinarski 2004).

It has been estimated that the population of Northern Quolls in Kakadu National Park prior to the arrival of the Cane Toad was approximately 80 000 individuals and that this population had probably declined by approximately 20% by the time Cane Toads had colonised approximately 20% of Kakadu National Park. It is expected that similar declines will occur in remaining Northern Quoll populations in the Northern Territory as the Cane Toad invades their habitat.

Western Australia

In Western Australia. the Northern Quoll is restricted to the Pilbara and Kimberley regions. Between 1900-1990, Northern Quolls had apparently disappeared from the south east and south west Kimberley region and had undergone a substantial decline in the Pilbara.

In the Pilbara region, the species distribution is now considered to be fragmented and mostly confined to the larger conservation reserves as well as to the Burrup Peninsula. Populations are considered to have been declining since the mid 1980s with the precise causes unknown. Surveys in the Kimberley in the 1970s showed that the Northern Quoll was widespread there. More recent surveys, undertaken during 2003 and early 2004, confirmed that the Northern Quoll was still numerous in the high rainfall coastal habitat of the west Kimberley. However, populations in the more arid east Kimberley region have apparently undergone a dramatic decline over the past 30 years. The reasons for these declines are also uncertain although it is considered that altered fire regimes and the impact of cattle (both feral and managed) on the landscape may have played an important role.

Currently, Cane Toads are absent from Western Australia and therefore have not contributed to a decline in numbers from the western part of the species' range. Other factors operating may include the impact of predation following increasing fire frequency and intensity. These factors are not well documented and the degree of decline is unknown. Ecoclimatic modelling has estimated that the Cane Toad will colonise the remainder of the Northern Quolls mainland range in Western Australia and it is predicted that this will occur within the next 10-20 years (Sutherst *et al*. 1995). While some populations in the west Kimberley may survive, as parts of this area may not be suitable for Cane Toads, the fate of the Northern Quoll is considered precarious in Western Australia as the Cane Toad will invade its habitat within the next few years.

Offshore Islands

The Northern Quoll is absent from Bathurst and Melville Islands but occurs on other smaller islands in the Northern Territory (Vanderlin, Marchinbar, Inglis, Groote and North-east) and has also been translocated to Astell and Pobasso islands. In Western Australia, the Northern Quoll is known from Augustus, Bigge, Boongaree, Caffarelli, Carlia, Dolphin, Hidden, Koolan, Sir Frederick, Uwins, and Wollaston islands. It is likely that the Cane Toad, which tolerates high levels of salinity, will colonise some offshore islands, including those closer to shore or with favourable tides, where Northern Quolls are known to occur.

Summary

Northern Quoll populations demonstrate a normal fluctuation in numbers involving a slight decline through the dry season. It is considered that the Northern Quoll is likely to, and will continue to, decline over most of its mainland, and some of its island, range. Based on population crashes during the 1990s in Cape York Peninsula and more recently from Kakadu National Park, and projecting a similar decline as the Cane Toad advances across the remainder of the Quoll's range, and even allowing for the persistence of isolated pockets in Queensland and in some offshore islands, this reduction has been estimated at about 95% of the range, and hence total species population, covering the period 1980 to 2010.

Over the last 10 years, the population has almost entirely been lost from the north east Top End, Northern Territory; Cape York Peninsula; and the Einasleigh Uplands of northern Queensland. These areas have been estimated to constitute approximately 30-40% of the Northern Quoll's pre-toad distribution. The viability of these remnant populations in the wild is, at this stage, unknown.

Over the next 10 years, the rest of the mainland Top End population is expected to also disappear, along with much of the Kimberley mainland population. These areas are estimated to constitute a further 30% of the species' pre-toad distribution. With the exception of some of its island locations, an almost total Cane Toad colonisation of the Northern Quolls range is expected.

While the actual reduction in total individuals can not be directly calculated, it is clear that a considerable contraction in range has occurred and that the loss of a number of populations has been observed in Queensland and in Kakadu National Park.

While Northern Quolls are still present in a number of localised areas in Queensland in which Cane Toads have been present for many years, they do not appear, to date, to be recolonising their former locations and to date there is little evidence that any substantive recovery has occurred following Cane Toad invasion. More recent experimental evidence from the Northern Territory supports anecdotal reports from Cape York Peninsula and other areas of Queensland that Northern Quoll populations have been, and continue to be, severely affected by the presence of Cane Toads.

Threats to the species continue to operate, including those that may have been instrumental in the background decline of the last fifty years or more. Most noticeably however, are the catastrophic declines being observed as a result of the lethal ingestion of Cane Toad toxin. As there is no known remedy, at this stage, to the advance of Cane Toads across the rest of the species range, the Northern Quoll can be expected to continue to decline commensurate with the historical decline in Queensland, and recent population crashes in Cape York Peninsula during the late 1980s to early 1990s and in Kakadu National Park over the last two years.

The Northern Quoll has undergone a substantial reduction in numbers, and is likely to continue to undergo in the immediate future, an ongoing reduction in numbers. While determining the extent of the species historic decline is difficult due to limited baseline data, it is conservatively estimated that the species past population size reduction is greater than 30%. It is considered that a number of island populations will remain free from Cane Toad invasion but that even given this, the likely extent and rate of the species future decline is predicted to be in the order of a further 30% as Cane Toads colonise the remainder of the species' mainland distribution.

TSSC acknowledges that, due to uncertainties over the impact of Cane Toads and the likely extent of the future decline in the species, the conservation status of the Northern Quoll falls somewhere between the vulnerable and endangered categories. Due to the continuing and present threat posed by Cane Toads, and the fact that over time this threat would also qualify the species for listing under Criterion 3, the TSSC considers that the Northern Quoll should be listed in the endangered category.

Therefore, the species is **eligible** for listing as **endangered** under this criterion.

Criterion 2 - Its geographic distribution is precarious for the survival of the species and is very restricted, restricted or limited.

The Northern Quoll once occurred continuously across northern Australia, from the Pilbara, Western Australia, to near Brisbane, Queensland. Over the last ten years the loss of populations from most of the north eastern Top End of the Northern Territory, Cape York Peninsula and the Einasleigh Uplands of northern Queensland has been observed, constituting approximately 30-40% of the Northern Quoll's distribution prior to the invasion of its habitat by the Cane Toad.

One study has estimated that a 75% reduction occurred in the Northern Quoll's range between 1900-1990 and that, during this period, the Northern Quoll was reduced to six major geographical centres: Drummond Range, central Queensland; wet tropics Northern Queensland; northern Cape York Peninsula; northern and western Top End Northern Territory; north Kimberley and Pilbara, Western Australia (Braithwaite and Griffiths 1994).

The Northern Quoll is likely to continue to disappear over most of its mainland, and some of its island, range. Based on evidence from Cape York Peninsula and more recently Kakadu National Park (and allowing for persistence in the existing pockets in Queensland and in some offshore islands), this reduction is estimated at about 95% of the range (and hence total population) as it was in 1980, by about 2010.

The pre-toad mainland distribution of the Northern Quoll in the Northern Territory is estimated to be 249 207 km^2. The species also occurs on offshore island, notably on the two main islands of Groote and Marchinbar, which contribute a further distribution of 2 487 km^2. By

1990 the species was considered to occupy approximately 87% of its former range (216 854 km²). Following invasion of the Cane Toad the Northern Territory mainland distribution is estimated to have declined to 20% of the species' 1990 distribution by 2004 (i.e. 47 812 km²).

Over the next 10 years, it is likely that the rest of the mainland Top End population will also disappear, as will much of the Kimberley mainland population, estimated to represent a further loss of approximately 30% of the species' pre-toad distribution.

A significant reduction in the species' range has already occurred, and is likely to continue. In addition the loss of a number of populations has been observed and the species has been reduced to a number of disjunct locations across northern Australia. However, there is insufficient quantitative data available to indicate that the geographic distribution of the Northern Quoll is precarious for the survival of the species. Therefore, the species is **not eligible** for listing under this criterion.

Criterion 3 - The estimated total number of mature individuals is limited to a particular degree and: (a) evidence suggests that the number will continue to decline at a particular rate; or (b) the number is likely to continue to decline and its geographic distribution is precarious for its survival.

Past anecdotal reports and more recent experimental field data suggest that the number of mature individuals will continue to decline. No baseline data appears to exist for the species in Queensland and there are no historical data on the species' abundance. There are no direct estimates on how limited the number of mature individuals is, or to what degree this number is continuing to decline. Conversely, there is no data indicating how viable surviving populations in Queensland are and there has been no research to date verifying that any recovery has, or is likely to occur.

The population of Northern Quolls in Kakadu National Park has been estimated to be in the order of 80 000 individuals, of which 20% is thought to have been lost to date following the invasion of the Cane Toad. There are no figures on the size of the Western Australian population or on those populations that are known to occur on offshore islands.

There is insufficient quantitative data available, at this stage, on the number of mature individuals and the geographic distribution of the species is not considered to be precarious for the survival of the species. Therefore, the species is **not eligible** for listing under this criterion.

Criterion 4 - The estimated total number of mature individuals is extremely low, very low or low.

There is no quantitative data available against this criterion. Therefore, the species is **not eligible** for listing under this criterion

Criterion 5 - Probability of extinction in the wild

There is no quantitative data available against this criterion. Therefore, the species is **not eligible** for listing under this criterion.

5. Conclusion

Northern Quoll populations have been in decline for some time due to a number of factors. The invasion of Cane Toads throughout their range has accelerated this decline to a rapid rate. Other threatening processes are considered to have been operating, and continue to operate on the species, notably in the western part of the species range.

The response of Northern Quoll populations to colonising Cane Toads indicates that the species is likely to experience a very significant decline in both range and abundance as Cane Toads continue to invade more of its habitat. It has been predicted that the Cane Toad will colonise all of the Northern Quoll's natural range, with the exception of some island populations, within the next 10-20 years.

While remnant populations are known to survive in Queensland, where Cane Toads have been present for many years, there is no research that indicates the viability of these populations in the wild in the longer term, or what conditions have led to their survival, or whether similar conditions may exist either in the Northern Territory or in Western Australia. To date, there is no evidence indicating that a recovery is occurring, or is likely to occur in populations in Queensland; or any evidence to suggest that Northern Quoll populations have, or will, survive in Cane Toad invaded regions of the Northern Territory.

The Northern Quoll has experienced a substantial decline over the last ten years and it is expected that it will experience a very severe decline over the next ten years as Cane Toads invade the remainder of its mainland habitat.

The species is **eligible** for listing as **endangered** under criterion 1.

6. Recommendation

TSSC recommends that the list referred to in section 178 of the EPBC Act be amended by including in the list in the endangered category:

- *Dasyurus hallucatus* (Northern Quoll)

Conservation Advice

Northern Quoll, *Dasyurus hallucatus*

The Northern Quoll is a solitary carnivorous marsupial, that makes its dens in rock crevices, tree holes or occasionally termite mounds, and is predominantly nocturnal. The species formerly occurred almost continuously and commonly across northern Australia from the Pilbara, Western Australia to near Brisbane, Queensland. It now occurs in a number of localised populations in Queensland, Northern Territory and Kimberley region (e.g. coastal Northern New South Wales, Queensland and Northern Territory, Western Australia NRM regions).

The Northern Quoll is absent from Bathurst and Melville Islands but occurs on other smaller islands in the Northern Territory (Vanderlin, Marchinbar, Inglis, Groote and North-east) and has also been translocated to Astell and Pobasso islands. In Western Australia, the Northern Quoll is known from Augustus, Bigge, Boongaree, Caffarelli, Carlia, Dolphin, Hidden, Koolan, Sir Frederick, Uwins, and Wollaston islands. It is likely that the Cane Toad, which tolerates high levels of salinity, will colonise some offshore islands, including those closer to shore or with favourable tides, where Northern Quolls are known to occur.

A number of factors are considered to be threatening the survival of the species:

- inappropriate fire regimes;
- predation following fire; and
- lethal toxic ingestion of Cane Toad toxin.

Poisoning as a result of the ingestion of Cane Toad toxin is considered to have had a catastrophic impact on a number of Northern Quoll populations. The Cane Toad will most likely colonise much of the remainder of the Northern Quoll's natural mainland range over the next 10-20 years, and it is likely that this decline will continue.

The priority recovery and threat abatement actions required for this species are to:

- minimise the impact of colonising Cane Toads on the species by:
  - investigating the use of physical barriers or other means, where feasible, to prevent the colonisation of key habitat areas;
  - undertaking translocation and management of Northern Quoll populations in safe havens where necessary;
- identifying areas of critical habitat (e.g. island populations);
- investigate the need to establish a captive breeding program for the species; and
- investigate the status of the species in Queensland, including the reasons for its survival following Cane Toad invasion.

This list does not encompass all actions that may be of benefit to this species, but highlights those that are considered to be of the highest priority at the time of listing.

A Recovery Plan is not yet in place for the Northern Quoll. A nomination to list 'Predation, competition and lethal toxic ingestion caused by Cane Toads, *Bufo marinus*' is currently under consideration by the Threatened Species Scientific Committee. The Committee will also, at the time it provides advice on listing of the nominated key threatening process, advise on whether or not a national Threat Abatement Plan would be a feasible, effective and efficient way to abate this process.

Priority for the development of recovery plan: High.

Publications used to assess the nomination

Braithwaite, R.W., and Griffiths, A. (1994). Demographic variation and range contraction in the northern quoll *Dasyurus hallucatus* (Marsupialia: Dasyuridae). *Wildlife Research* **21**:203-17.

Burnett, S. (1997). Colonising cane toads cause population declines in native predators: reliable anecdotal information and management implications. *Pacific Conservation Biology* **3**:65-72.

Oakwood, M. (2000). Reproduction and demography of the northern quoll, *Dasyurus hallucatus*, in the lowland savanna of northern Australia. *Australian Journal of Zoology* **48**, 519-539.

Oakwood, M. (2004). The Effect of Cane Toads on a Marsupial Carnivore, the Northern Quoll, *Dasyurus hallucatus*. Report to Parks Australia.

Sutherst, R.W., Floyd, R.B., and Maywald, G.F. (1995). The potential distribution of the cane toad, *Bufo marinus* L. in Australia. *Conservation Biology* **10**, 294-299.

van Dam, R.A., Walden, D.J., and Begg, G.W. (2002). A preliminary risk assessment of cane toads in Kakadu National Park. Report 164.

Watson, M. and Woinarski, J. (2003). A preliminary assessment of impacts of cane toads on terrestrial vertebrate fauna in Kakadu National Park.

Watson, M., and Woinarski, J. (2003). Vertebrate monitoring and re-sampling in Kakadu National Park, 2002. Report to Parks Australia, Tropical Savannas Cooperative Research Centre, Darwin.

Watson, M., and Woinarski, J. (2004). Vertebrate monitoring and re-sampling in Kakadu National Park, Year 3, 2003-04. Report to Parks Australia, Tropical Savannas Cooperative Research Centre, Darwin.

Woinarski, J.C.Z., Milne, D.J., and Wanganeen, G. (2001). Changes in mammal populations in relatively intact landscapes of Kakadu National Park, Northern Territory, Australia. *Austral Ecology* **26**:360-370.

Woinarski, J.C.Z., Risler, J., and Kean, L. (2004). The response of vegetation and vertebrate fauna to 23 years of fire exclusion in a tropical Eucalyptus open forest, Northern Territory, Australia. *Austral Ecology* **29**, 156-176.

The Minister approved this Conservation Advice and transferred this species from the Endangered to the Vulnerable category, effective from 07/12/2016

Conservation Advice

Hipposideros semoni

Semon's leaf-nosed bat

Note: The information contained in this conservation advice was primarily sourced from 'The Action Plan for Australian Mammals 2012' (Woinarski et al., 2014). Any substantive additions obtained during the consultation on the draft have been cited within the advice. Readers may note that conservation advices resulting from the Action Plan for Australian Mammals show minor differences in formatting relative to other conservation advices. *These reflect the desire to efficiently prepare a large number of advices by adopting the presentation approach of the Action Plan for Australian Mammals, and do not reflect any difference in the evidence used to develop the recommendation.*

Taxonomy

Conventionally accepted as *Hipposideros semoni* (Matschie 1903).

No subspecies are currently recognised. Semon's leaf-nosed bat is closely related to several *Hipposideros* species in northern Australia, such as *H. stenotis* (northern leaf-nosed bat) and in New Guinea (Hill 1963), such as *H. muscinus* (Fly River leaf-nosed bat). A current taxonomic study is comparing closely related forms in Australia and in New Guinea (Armstrong pers. comm., cited in Woinarski et al., 2014).

Summary of assessment

Conservation status

Vulnerable: Criterion 2 B2(a),(b)(iii) and Criterion 3 C2(a)(i)

Semon's leaf-nosed bat was listed as Endangered under the EPBC Act in 2001. Following a formal review of the listing status of Semon's leaf-nosed bat, the Threatened Species Scientific Committee (the Committee) has determined that there is sufficient evidence to support a change of status of the species under the EPBC Act from Endangered to Vulnerable.

Species can be listed as threatened under state and territory legislation. For information on the listing status of this species under relevant state or territory legislation, see http://www.environment.gov.au/cgi-bin/sprat/public/sprat.pl

Reason for conservation assessment by the Threatened Species Scientific Committee

This advice follows assessment of new information provided to the Committee to reassess the listing status of *Hipposideros semoni.*

Public Consultation

Notice of the proposed amendment and a consultation document was made available for public comment for 32 business days between 29 February 2016 and 15 April 2016. Any comments received that were relevant to the survival of the species were considered by the Committee as part of the assessment process.

Species Information

Description

Semon's leaf-nosed bat is a small bat with a head to body length of approximately 40−50 mm and a weight of 6−10 g (Hall 2008). The fur is relatively long and has a ruffled appearance; it is dark smoky-grey in colour, though lighter on the belly (Churchill 1998, 2009). The wing membrane near the body is covered with whitish-brown hair (Hall 2008). The ears are particularly long and narrow, with an acute point (Churchill 1998).

The noseleaf is well developed, square-shaped and covers most of the muzzle. There are two wart-like protuberances - one in the centre and another on the posterior edge (Churchill 1998, 2009). The upper portion of the noseleaf is divided into four depressions and there are two supplementary leaflets under each side of the lower portion (Hall 2008). It is distinguished from *Hipposideros stenotis* (northern leaf-nosed bat) by having a longer central wart (Hall 2008).

Distribution

Semon's leaf-nosed bat occurs mainly in north-eastern Australia (along eastern Cape York Peninsula to Townsville), with the majority of records around Iron Range, Kulla, Oyala Thumotang and Cape Melville National Parks, and near Cooktown (Reardon et al., 2010). There is evidence for an isolated subpopulation further south at Kroombit Tops (south of Gladstone) (Schulz & de Oliveira 1995). Beyond Australia, it is also known from a few records in New Guinea (Flannery 1990, 1995; Bonaccorso 1998).

Bonaccorso et al. (2008) reported that the range of the species has receded northwards considerably (by approximately 30 percent of its Australian range) over the last 60 years.

Relevant Biology/Ecology

Semon's leaf-nosed bat is a poorly-known and rare species (Bonaccorso et al., 2008; Woinarski et al., 2014), which probably occurs in low densities even within core habitats (Armstrong pers. comm., 2016). It mainly occurs in rainforests, but has also been recorded from streams and rivers adjacent to rainforest (Reardon et al., 2010). A wide range of roost sites have been recorded, including in houses (Van Deusen 1975), abandoned buildings (Churchill 2009), caves (Thomson et al., 2001; Churchill 2009) and trees (Churchill 2009). Semon's leaf-nosed bat has short broad wings, and its flight is typically slow and fluttering, usually within two metres of the ground (Van Deusen 1975; Hall 2008). It is insectivorous; moths may comprise the main dietary item (Churchill 2009).

Semon's leaf-nosed bat is sexually dimorphic. Females are larger than males (Whybird et al., 1998) and echolocation call frequency varies between the sexes (de Oliveira & Schulz 1997; Armstrong pers. comm., cited in Woinarski et al., 2014). Males produce a constant frequency call of approximately 94 kHz and females produce a constant frequency call of approximately 74 kHz. Calls of this species have also been noted in the 83-85 kHz band. These characteristic differences may reflect or drive sexual differences in foraging and diet (Whybird pers. comm., cited in Woinarski et al., 2014; Armstrong pers. comm., cited in Woinarski et al., 2014).

Females give birth to a single young per year, around November (Churchill 2009). A generation length of 6-7 years is derived from a mean of age at sexual maturity (estimated at 1-2 years) and longevity (probably around 12 years), based on information for other *Hipposideros* species. No detailed information is available for this species.

Semon's leaf-nosed bat is difficult to catch while foraging as its slow flight and manoeuvrability allows it to avoid nets; however, it has a distinctive echolocation call.

Threats

Threats to Semon's leaf-nosed bat are outlined in the table below (Woinarski et al., 2014). Habitat loss and degradation (due to clearing, inappropriate fire regimes and other human activities) is postulated to be the key threat to the species.

Threat factor	Consequence rating	Distributional extent over which threat may operate	Evidence base
Disturbance, destruction or reduced accessibility to roost sites	Moderate	Minor	Thompson et al. (2001) regarded disturbance, destruction and reduced accessibility to roost sites by human visitation and mining a plausible threat to the species. This threat, however, has not been demonstrated.
Habitat loss and fragmentation	Moderate	Minor	Woinarski et al. (2014) consider habitat loss and fragmentation to be a possible threat to the species. This threat, however, has not been demonstrated.
Habitat change due to pastoralism	Minor	Moderate	Dennis (2012) considered habitat change as a result of pastoralism to be a possible threat to the species. This threat, however, has not been demonstrated.
Increased fire extent, frequency and intensity	Minor	Moderate	The species range is located over areas of differing fire regime. Dennis (2012) considered extensive, frequent and intense fires to be a possible threat to the species due to the impacts on prey abundance. This threat, however, has not been demonstrated.
Predation by cats (*Felis catus*)	Minor	Minor	Woinarski et al. (2014) consider predation by cats at roost sites and roost entrances to be a possible threat to the species. This threat, however, has not been demonstrated.

How judged by the Committee in relation to the EPBC Act Criteria and Regulations

Criterion 1. Population size reduction (reduction in total numbers) Population reduction (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

Evidence:

Insufficient data to determine eligibility

Population trends are difficult to ascertain for low density species, and there has been little survey effort in recent decades. The Semon's leaf-nosed bat population may be declining according to Duncan et al. (1999), based in part on comparison of relative numbers reported in the 1990s and during surveys of the Cape York Peninsula in the 1940s and 1950s (Tate 1952). There are no data available on why a decline may have occurred (Dennis 2012).

In more recent surveys, Reardon et al. (2010) noted that the species was regularly reported in their targeted searches of Cape York Peninsula. They considered that the species is relatively secure within the Cape York Peninsula portion of its range and the assumption of a decline may not be valid. In addition, rainforest in parts of the Iron Range area has expanded over recent decades due to the current fire regime (Russell-Smith et al., 2004).

However, over the last 60 years the range of the species in Australia has receded northwards by approximately 30 percent (Bonaccorso et al., 2008). Preliminary modelling has predicted that the species' preferred habitat of rainforest and riparian forest is likely to reduce in area over the next 50 years, particularly in the south of its range (Inkster pers. comm., cited in Woinarski et al., 2014).

Woinarski et al. (2014) consider that, if a decline in population size is occurring, it is likely to be at a rate of less than 30 percent over a 20 year period (approximately three generations).

The Committee considers that there is insufficient information to determine the eligibility of the species for listing in any category under this criterion. A decline is probably occurring but there are no data to demonstrate the rate of decline.

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 subpopulations; (v) number of mature individuals			
(c) Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations; (iv) number of mature individuals			

Evidence:

Eligible under Criterion 2 B2(a),(b)(iii) for listing as Vulnerable

The extent of occurrence is estimated at 162 008 km^2, and the area of occupancy is estimated at 128 km^2. These figures are based on the mapping of point records from 1976 to 2016, obtained from state governments, museums and CSIRO. The extent of occurrence was calculated using a minimum convex hull, and the area of occupancy calculated using a 2x2 km grid cell method, based on the IUCN Red List Guidelines 2014 (DotE 2015). Woinarski et al. (2014) considered that the calculated AOO, which they estimated to be 108 km^2, is a significant under-estimate due to limited sampling across the occupied range, and may be as high as 2000 km^2. Robust estimates of EOO and AOO are not possible due to insufficient survey effort.

However, the available information suggests that the AOO is limited or restricted, as it likely lies between 128 km^2 and 2000 km^2. There is a continuing decline in the area and extent of habitat (see Criterion 1). The species is not severely fragmented, and is present at more than 5 locations (Woinarski et al., 2014) but probably fewer than 10 locations. This indicates that the species likely meets the thresholds for Vulnerable, but not Endangered.

The Committee considers that the species meets the relevant elements of Criterion 2 to make it eligible for listing as Vulnerable.

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 subpopulation	≤ 50	≤ 250	≤ 1,000
(a) (ii) % of mature individuals in one subpopulation =	90 – 100%	95 – 100%	100%
(b) Extreme fluctuations in the number of mature individuals			

Evidence:

Eligible under Criterion 3 C2(a)(i) for listing as Vulnerable

There are no robust estimates of population size or the size of subpopulations. The previous EPBC Act listing was based on a population size estimate of fewer than 2500 mature individuals. Several population estimates have been made since this decision:

- Based on more recent surveys at the Iron Range and McIlwraith Range regions on Cape York Peninsula, Reardon et al. (2010) considered that this figure is likely to be an underestimate and that it is likely 'the population...exceeds 2500', although it is 'not abundant.'
- Woinarski et al. (2014) consider that the population size of Semon's leaf-nosed bat is likely to be greater than 10 000 mature individuals, and the largest subpopulation is likely to contain less than 1000 mature individuals. They consider that the population may be declining, but probably at a rate of less than 10 percent over a three generation period. There is no information to suggest there have been extreme fluctuations in the number of mature individuals.
- The Australasian Bat Society Inc. (pers. comm., 2016) and K. Armstrong (pers. comm., 2016) consider that the population size of Semon's leaf-nosed bat is likely to be less than 10 000 mature individuals, given that the species would probably utilise a relatively small proportion of habitat in its distribution, and that it occurs at low densities.

The Committee considers that the estimated total number of mature individuals of this species is likely to be between 2500 and 10 000 (i.e. limited), there is an inferred continuing decline in numbers (see Criterion 1), and the geographic distribution is precarious for the survival of the species as the number of individuals in each subpopulation is likely to be less than 1000. Therefore, the species meets the relevant elements of Criterion 3 to make it eligible for listing as Vulnerable.

Criterion 4. Number of mature individuals			
	Critically Endangered Extremely low	Endangered Very Low	Vulnerable Low
Number of mature individuals	< 50	< 250	< 1,000

Evidence:

Not eligible

The population is likely to be larger than 2500 based on all current population estimates. Therefore, the species does not meet this required element of this criterion.

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

Evidence:

Insufficient data to determine eligibility

Population viability analysis has not been undertaken.

Conservation Actions

Recovery Plan

A multi-species recovery plan is currently in place for three species of cave-dwelling bats including Semon's leaf-nosed bat. The *Recovery plan for cave-dwelling bats, Rhinolophus philippinensis, Hipposideros semoni and Taphozous troughtoni 2001–2005* (Thompson et al., 2001) was developed by the State of Queensland and adopted as a national recovery plan under the EPBC Act in 2007.

The recovery plan includes the following objectives:

- establish the status of poorly known species and to identify appropriate species management units within two years of implementation of the plan;
- gather the necessary biological data from current records and through new, targeted field work for the effective conservation management of the species;
- implement conservation strategies or on-ground conservation works in priority sites where the species occur to mitigate identified threatening processes; and
- identify trends in the species' abundance at priority sites across their distributional ranges after the instigation of conservation strategies or on-ground conservation works.

Previous and current studies, particularly regarding the biology of the species, have contributed towards meeting the objectives of the plan since its adoption (e.g. Reardon et al., 2010). However, further research is required to establish population trends, clarify threatening processes and develop appropriate management actions. The plan is scheduled to cease in 2017.

The Committee recommends that the existing recovery plan not be renewed after it ceases in 2017, as its continuation would not add significant benefit above an approved Conservation Advice. This Conservation Advice provides sufficient direction to implement priority actions, mitigate key threats and enable recovery of the species.

Primary Conservation Actions

1. Undertake targeted surveys to identify important subpopulations, roost sites and habitat requirements.
2. Protect all roost sites and important subpopulations.
3. Maintain the quality of habitat, particularly at roost sites.
4. Assess population size, trends in population and distribution, and the relative impacts of threats.

Further habitat loss from activities such as land clearing and mining, in areas which contain roost sites or important subpopulations, is likely to have a significant impact on the species. Prior to any clearing or development within the subspecies' distribution, targeted surveys for Semon's leaf-nosed bat should be undertaken.

Conservation and Management Actions

There are no specific management actions targeted at this species. Parts of its range are in conservation reserves where some threats are managed. There has been some management of abandoned mines within the species' range (Thomson 2002), but such actions are constrained by limited information about the roost preferences and locations of this species.

Recommended conservation and management actions are outlined in the table below (Woinarski et al., 2014).

Theme	Specific actions	Priority
Active mitigation of threats	Constrain actions that may lead to loss of critical roost sites.	High
	If needed, stabilise roost sites; and minimise disturbance.	Medium
	Reduce the frequency, extent and intensity of controlled burns.	Low-medium
	Implement broad-scale management of feral cats; or local-scale implementation at and around important subpopulations.	Low
Captive breeding	N/a	
Quarantining isolated populations	N/a	
Translocation	N/a	
Community engagement	Involve Indigenous ranger groups in survey, monitoring and management.	Medium
	Collaborate with landholders and other stakeholders to prevent loss and disturbance of roost sites.	Medium

Survey and monitoring priorities

Theme	Specific actions	Priority
Survey to define better distribution	Undertake fine-scale sampling to assess distribution and identify and circumscribe important subpopulations (or colonies) (and roost sites), and assess the population size of these.	High

Establish or enhance monitoring program	Design an integrated bi-annual monitoring program across its range (including at known roost sites) to determine population trends; surveys should be undertaken in both the wet and dry seasons.	Medium-high
	Implement an integrated monitoring program linked to an assessment of management effectiveness.	Medium-high

Information and research priorities

Theme	Specific actions	Priority
Assess relative impacts of threats	Assess the structural viability of all known roost sites.	Medium
	Assess potential threats (particularly human visitation) to all known roost sites.	Medium
	Identify the population-level responses to a range of fire regimes, and model population viability across all fire scenarios.	Medium
	Assess abundance of feral cats in the range of this species, and the impact of predation on population viability.	Low
Assess effectiveness of threat mitigation options	Assess options for gating or other management of roost sites.	Medium
	Assess efficacy and impacts of management options to reduce fire incidence, extent and intensity.	Medium
	Assess effectiveness of options for broad-scale control of feral cats; or of local scale control at sites with important populations.	Low
Resolve taxonomic uncertainties	Establish genetic structuring across subpopulations to identify extent of movement of individuals, and to identify populations that may be most genetically distinctive.	Medium
Assess habitat requirements	Characterise roosting requirements, including maternity and non-breeding roosts.	Medium
	Investigate seasonal and spatial patterning of foraging habitat use (of both sexes).	Low
Assess diet, life history	Assess the extent to which food availability may be affected by fire regimes.	Medium
	Investigate key components of diet (for both sexes).	Low

Recommendations

(i) The Committee recommends that the list referred to in section 178 of the EPBC Act be amended by **transferring** from the Endangered category to the Vulnerable category:

Hipposideros semoni

(ii) The Committee recommends that there not be a recovery plan for the species.

Threatened Species Scientific Committee

06/09/2016

References cited in the advice

Bonaccorso, F. J. (1998). *Bats of Papua New Guinea*. Conservation International Tropical Field Guide Series. Conservation International, Washington, D.C.

Churchill, S.K. (1998). *Australian Bats*. Reed New Holland, Sydney.

Churchill, S. K. (2009). *Australian Bats*. Second edition. Allen and Unwin: Crows Nest.

Dennis, A. J. (2012). Semon's leaf-nosed bat, *Hipposideros semoni*. In *Queensland's threatened animals* (eds L. K. Curtis, A. J. Dennis, K. R. McDonald, P. M. Kyne & S. J. S. Debus), pp. 378-379. CSIRO Publishing, Collingwood.

de Oliveira M. C., & Pavey, C.R. (1995). In search of *Hipposideros semoni* at St Mary's State Forest, south-east Queensland. *The Australasian Bat Society Newsletter 4*, 46-48.

de Oliveira M. C., & Schulz, M. (1997). Echolocation and roost selection in Semon's leaf-nosed bat *Hipposideros semoni*. *Memoirs of the Queensland Museum 42*, 158.

Department of the Environment (DotE) (2015). *Area of Occupancy and Extent of Occurrence for* Hipposideros semoni. Unpublished report, Australian Government Department of the Environment, Canberra.

Duncan, A., Baker, G. B., & Montgomery, N. (1999). *The action plan for Australian bats*. Environment Australia, Canberra.

Flannery, T. F. (1990). *The mammals of New Guinea*. Robert Brown, Carina.

Flannery, T. F. (1995). *Mammals of New Guinea*. Revised and updated edition. Reed Books, Chatswood.

Hall, L. S. (2008). Semon's leaf-nosed bat *Hipposideros semoni*. In *The mammals of Australia*. Third edition. (Eds S. Van Dyck & R. Strahan), pp. 465-466. Reed New Holland, Sydney.

Hill, J. E. (1963). A revision of the genus *Hipposideros*. *Bulletin of the British Museum (Natural History), Zoology 11*, 1-129.

Reardonl.2010). *Review of the threatened status of microchiropteran bat species on Cape York Peninsula.* South Australian Museum, Adelaide.

Russell-Smith, J., Stanton, P. J., Whitehead, P .J., & Edwards, A. (2004). Rainforest invasion of eucalypt-dominated woodland savanna, Iron Range, north-eastern Australia II. Rates of landscape change. *Journal of Biogeography 31*, 1305-1316.

Schulz, M., & de Oliveira, M.C. (1995). Microchiropteran fauna of Kroombit Tops, central Queensland, including a discussion on survey techniques. *Australian Zoologist 30*, 71-77.

Tate, G. H. H. (1952). Results of the Archbold Expeditions. No. 66. Mammals of Cape York Peninsula, with notes on the occurrence of rain forest in Queensland. *Bulletin of the American Museum of Natural History 98*, 563-616.

Thomson, B. (2002). *Australian handbook for the conservation of bats in mines and artificial cave-bat habitats.* Australian Centre for Mining Environmental Research, Melbourne.

Thomson, B., Pavey, C., & Reardon, T. (2001). *National recovery plan for cave-dwelling bats,* Rhinolophus philippensis, Hipposideros semoni *and* Taphozous troughtoni *2001-2005.* Queensland Environmental Protection Agency, Brisbane.

Van Deusen, H. M. (1975). History of Semon's Horseshoe Bat in Australia. *The North Queensland Naturalist 42*, 4-5.

Whybird, O. J., Coles, R. B. & Clague, C. I. (1998). Sexual dimorphism and the echolocation of *Hipposideros semoni.* Abstracts from the 8th Australasian Bat Society Conference, Rockhampton, 1998, p. 9.

Woinarski, J. C. Z., Burbidge, A. A., & Harrison, P. L. (2014). *The Action Plan for Australian Mammals 2012*. CSIRO Publishing, Collingwood.

Other sources cited in the advice

Armstrong, K. (2016). Comment on the conservation listing re-assessments for *Hipposideros semoni*, *Rhinolophus robertsi* and *Saccolaimus saccolaimus*. Received 15 April 2016. Zoologist and IUCN Bat Specialist Group member.

Australasian Bat Society Inc. (2016). Comment on listing re-assessments for *Hipposideros semoni*, *Rhinolophus robertsi* and *Saccolaimus saccolaimus*. Received 15 April 2016.

Bonaccorso, F., Hall, L., & Thomson, B. (2008). *Hipposideros semoni*. In 'The IUCN Red List of Threatened Species.' Version 2011.2. Viewed 23 May 2012. Available on the internet at: http://www.iucnredlist.org.

The Minister approved this conservation advice on 27/10/2015 and agreed that this species should retain its current listing status of vulnerable under the EPBC Act

Conservation Advice

Geophaps scripta scripta

squatter pigeon (southern)

Taxonomy

Conventionally accepted as *Geophaps scripta scripta* (Temminck, 1821). The squatter pigeon (southern) is one of two subspecies, the other being *Geophaps scripta peninsulae* (squatter pigeon (northern)).

Summary of assessment

Conservation status

Vulnerable

The squatter pigeon (southern) was transferred from the *Endangered Species Protection Act 1992* (ESP Act) to the Vulnerable list of the *Environmental Protection and Biodiversity Conservation Act* (1999) (EPBC Act) when the latter came into force in July 2000. For a species to be considered as Vulnerable under the ESP Act, the Minister must have been satisfied that the species was likely to become endangered within the next 25 years.

Following a formal review of the listing status of the squatter pigeon (southern), the Threatened Species Scientific Committee (the Committee) has determined that there is no evidence that the species has undergone any demonstrable recovery since being listed; and that there is insufficient evidence to support a change of status under the EPBC Act. Therefore, the Committee concluded that the squatter pigeon (southern) should remain listed as Vulnerable under the EPBC Act.

Species can be listed as threatened under state and territory legislation. For information on the listing status of this species under relevant state or territory legislation, see http://www.environment.gov.au/cgi-bin/sprat/public/sprat.pl

Reason for conservation assessment by the Threatened Species Scientific Committee

This advice follows assessment of new information provided to the Committee to re-assess the listing status of *Geophaps scripta scripta,* for potential de-listing.

Relevant part of the EPBC Act for amending the list of threatened native species

Section 186 of the EPBC Act states that:

"(2A) The Minister must not delete (whether as a result of a transfer or otherwise) a native species from a particular category unless satisfied that:

(a) the native species is no longer eligible to be included in that category; or

(b) the inclusion of the native species in that category is not contributing, or will not contribute, to the survival of the native species."

Public Consultation

Notice of the proposed amendment and a consultation document was made available for public comment for greater than 30 business days between 17 November 2014 and 9 January 2015.

Any comments received that were relevant to the survival of the species were considered by the Committee as part of the assessment process.

Species/Subspecies Information

Description

The squatter pigeon (southern) is a medium-sized, ground-dwelling bird approximately 30 cm in length and weighing 190–250 g. Adults are predominantly grey-brown, with bold black and white stripes on the face and throat. The upperwings are dark-brown, the upperbreast light grey-brown grading to blue-grey on the lower breast and centre of the belly, and the rest of the belly and flanks are white. The underwings are white with a dark leading edge. It has a black bill and dull-purple legs and feet. The sexes are similar in appearance. Juveniles can be distinguished from adults by their duller colouring and less distinctive black and white facial stripes (Higgins & Davies, 1996).

The southern and northern subspecies of the squatter pigeon appear virtually identical, except that the southern subspecies is slightly larger, and the skin around the eyes is predominantly blue-grey compared to yellowy-orange to orange-red in the northern subspecies (Ford, 1986; Higgins & Davies, 1996).

Distribution

The squatter pigeon (southern) occurs on the inland slopes of the Great Dividing Range. Its current distribution extends from the Burdekin-Lynd Divide in central Queensland, west to Longreach and Charleville, east to the coast between Port Curtis and Proserpine, and south to New South Wales (NSW) north of 29° S (Cooper et al., 2014). There is a broad zone of hybridisation with the northern subspecies along the Burdekin-Lynd Divide (Higgins & Davies, 1996; Garnett & Crowley, 2000).

The subspecies has disappeared from the southern half of its historical range. Formerly widespread and abundant in NSW, occurring south to West Wyalong at 34°S, its range has contracted markedly since the 1870s. There have been few sightings in NSW since 1975, with only three confirmed reports since 2000 (Higgins & Davies, 1996; Garnett & Crowley, 2000; Cooper et al., 2014).

Relevant Biology/Ecology

The squatter pigeon (southern) inhabits the grassy understorey of open eucalypt woodland, and less often savannas. It is nearly always found near permanent water such as rivers, creeks and waterholes. Sandy areas dissected by gravel ridges, which have open and short grass cover allowing easier movement, are preferred. It is less commonly found on heavier soils with dense grass. It often occurs in burnt areas and is sometimes found on tracks and roadsides (Higgins & Davies, 1996; Garnett & Crowley, 2000).

The subspecies nests on the ground, usually laying two eggs among or under vegetation. It forages for seeds among sparse and low grass, in improved pastures, and beside railway lines and with domestic fowl around settlements. It roosts in low trees at night. Its movements are poorly known but it appears to be locally dispersive or resident, with no long-distance seasonal movements recorded (Higgins & Davies, 1996). The generation time is estimated at 5 years (Garnett & Crowley, 2000).

Threats

The population declined rapidly during the late 19th and early 20th centuries, and continued to decline in NSW and southern Queensland where it is now very rare (Cooper et al., 2014). In NSW, the disappearance of the subspecies has been attributed to overgrazing at times of drought, followed by clearing of vegetation. Its original habitat in NSW is nearly all now grazed by sheep or is under cultivation. In Queensland, much of its original habitat has been replaced with improved pasture for cattle-grazing which, while decreasing the abundance of natural food plants, is not as destructive as grazing by sheep and may provide an important source of food (Higgins & Davies, 1996; Garnett & Crowley, 2000).

Current threats include ongoing vegetation clearance and fragmentation, overgrazing of habitat by livestock and feral herbivores such as rabbits (*Oryctolagus cuniculus*), introduction of weeds, inappropriate fire regimes, thickening of understorey vegetation, predation by feral cats (*Felis catus*) and foxes (*Vulpes vulpes*), trampling of nests by domestic stock and illegal shooting (Garnett & Crowley, 2000; Stewart, pers. comm. 2015).

How judged by the Committee in relation to the EPBC Act Criteria and Regulations

Criterion 1. Population size reduction (reduction in total numbers) Population reduction (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

Evidence:

Insufficient data to determine eligibility

The available information suggests that the squatter pigeon (southern) has continued to decline in southern Queensland and northern NSW. However, it is unclear how much of this decline occurred over the past three generations (15 years), as sub-populations in these regions are very low and there are insufficient data to determine trends. In Queensland, small colonies that once persisted in the south-eastern region are no longer found, with only one record in the Toowomba-Lockyer area and one record near Sundown National Park in the past few years (Stewart, pers. comm. 2015).

In NSW there were no confirmed reports between 1980 and 2000, and only three confirmed reports since 2000 (Cooper et al., 2014; Cooper, pers. comm. 2015). Breeding has not been recorded in NSW at any time during the past 50 years, suggesting that there is little or no remaining suitable breeding habitat (Cooper et al., 2014; Cooper, pers. comm. 2015). The NSW population is estimated to be extremely low at <100 individuals with an extent of occurrence estimated at <1000 km^2 (Cooper et al., 2014; Cooper, pers. comm. 2015).

The subspecies remains common north of the Carnarvon Ranges in central Queensland, where it is likely distributed as a single, continuous (i.e. inter-breeding) sub-population (Squatter

Pigeon Workshop, 2011). Numerous, recent records of the subspecies in the region between Injune and the Carnarvon Ranges (QLD DEHP, 2012) suggest that squatter pigeons (southern) found in this region are also part of the northern, continuous sub-population. However, no surveys have been undertaken in central Queensland to determine its status, and threatening processes such as fire, vegetation thickening, and coal and gas mining are likely to be affecting its habitat (Stewart, pers. comm. 2015).

Following assessment of the available information the Committee has determined that while there is evidence of ongoing population declines, the available evidence is insufficient to determine whether the rate of decline has changed during recent decades. Therefore, the Committee has determined there is insufficient data to judge whether the status of the squatter pigeon (southern) against this criterion should be changed from its current Vulnerable listing.

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 km^2**	**< 5,000 km^2**	**< 20,000 km^2**
B2. Area of occupancy (AOO)	**< 10 km^2**	**< 500 km^2**	**< 2,000 km^2**
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 subpopulations; (v) number of mature individuals			
(c) Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations; (iv) number of mature individuals			

Evidence:

Not eligible

Garnett & Crowley (2000) estimated the extent of occurrence to be 440 000 km^2 and the area of occupancy to be 10 000 km^2. These estimates were considered to be of medium and low reliability respectively. There are no other estimates available describing the species extent or area of occupancy.

The Queensland Resources Council (2015), which used GIS data (based on a 2 x 2 km grid) to examine the extent of squatter pigeon (southern) habitat overlapping resource industry sites, calculated the extent of occurrence to be 1 684 230 km^2 and the area of occupancy to be 2 888 km^2. Although it is unclear whether this entire habitat area is presently occupied, these data only covered a proportion of the subspecies' potential habitat, and support the conclusion that it does not meet the thresholds under Criterion B1 and B2.

Following assessment of the information the Committee has determined that the geographic distribution is not limited. Therefore, the subspecies has not been demonstrated to have met this required element of this criterion.

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 generations (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 subpopulation	≤ 50	≤ 250	≤ 1,000
(a) (ii) % of mature individuals in one subpopulation =	90 – 100%	95 – 100%	100%
(b) Extreme fluctuations in the number of mature individuals			

Evidence:

Not eligible

Garnett & Crowley (2000) estimated the number of mature individuals to be approximately 40 000, although this was considered to be of low reliability. Limited surveys have been undertaken and there are no reliable estimates of current population size or trends (Stewart, pers. comm. 2015); however, given the previous population estimate it is unlikely that the species meets the threshold for listing under this criterion.

The Committee considers that the subspecies is ineligible for listing under any category in this criterion as it is thought there are likely to be more than 10,000 mature individuals in the population.

Criterion 4. Number of mature individuals			
	Critically Endangered Extremely low	Endangered Very Low	Vulnerable Low
Number of mature individuals	< 50	< 250	< 1,000

Evidence:

Not eligible

Garnett & Crowley (2000) estimated the number of mature individuals to be approximately 40 000. Although this estimate was considered to be of low reliability and is out of date, it is highly unlikely that the current population is <1000 mature individuals.

The estimated number of mature individuals is approximately 40 000 which is not considered extremely low, very low or low. Therefore, the species has not been demonstrated to have met this required element of this criterion.

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**

Evidence:

Insufficient data to determine eligibility

Population viability analysis has not been undertaken.

Consideration for delisting

The assessment indicates that there is insufficient evidence to judge whether the squatter pigeon (southern) is no longer eligible to be listed as Vulnerable under the EPBC Act. There is evidence that declines are continuing in the southern part of its range, and considerable uncertainty regarding population trends across its total range due to insufficient survey effort. It cannot be clearly demonstrated that the subspecies is ineligible for listing under Criterion 1.

The inclusion of the squatter pigeon (southern) in the Vulnerable category is contributing to the survival of the subspecies, as the EPBC Act requires project proponents to refer a proposal for assessment if it may have a significant impact on a threatened species. Where appropriate, the department has issued conditions requiring proponents to avoid, minimise or mitigate impacts on the subspecies.

Conservation Actions

Recovery Plan

The Committee recommends that there should not be a recovery plan for *Geophaps scripta scripta* (squatter pigeon (southern)) as the approved conservation advice for the subspecies provides sufficient direction to implement priority actions and mitigate against key threats.

Primary Conservation Action

Conservation and Management Actions

- Identify sub-populations of high conservation priority, especially in the southern part of the squatter pigeon’s (southern) range.
- Protect and rehabilitate areas of vegetation that support important sub-populations.
- Protect sub-populations of the listed subspecies through the development of covenants, conservation agreements or inclusion in reserve tenure.
- Develop and implement a stock management plan for key sites.
- Develop and implement a management plan, or nominate an existing plan to be implemented, for the control and eradication of feral herbivores in areas inhabited by the squatter pigeon (southern).
- Raise awareness of the squatter pigeon (southern) within the local community, particularly among land managers.

Survey and Monitoring priorities

- Monitor selected sub-populations throughout the distribution of the subspecies to identify rates of population change.

Information and Research priorities

- Identify preferred food plants, and the responses of these to fire and grazing regimes.
- Determine patterns of dispersal or residency, and the factors that may determine these.
- Assess reproductive success, and the factors that affect this.
- Assess the species' status, and the impacts of mining, in central Queensland.

Recommendations

(i) The Committee recommends that *Geophaps scripta scripta* should retain its current listing status of Vulnerable under the EPBC Act as there is insufficient evidence to support transferring it to a different category.

(ii) The Committee recommends that there should not be a recovery plan for this subspecies.

Threatened Species Scientific Committee

02/09/2015

References cited in the advice

ALA - OEH (2006). Atlas of Living Australia. NSW Office of Environment and Heritage.

Blakers M, Davies SJJF and Reilly PN (1984). *The Atlas of Australian Birds*. Melbourne University Press.

Cooper, RM (2015). Personal communication by email, 13 April 2015. New South Wales Bird Atlassers Inc.

Cooper RM, McAllan IAW and Curtis BR (2014). *The Atlas of the Birds of NSW and the ACT*. Mini-Publishing, Gordon, New South Wales.

Ford J (1986). Avian hybridisation and allopatry in the region of the Einasleigh Uplands and Burdekin-Lynd Divide, north-eastern Queensland. *Emu* 86**,** 87–110.

Garnett ST and Crowley GM (2000). *The Action Plan for Australian Birds 2000.* Environment Australia, Canberra.

Garnett ST, Szabo JK and Dutson G (2011). *The Action Plan for Australian Birds 2010*. Birds Australia, CSIRO Publishing, Melbourne.

Higgins PJ and Davies SJJF (1996). *Handbook of Australian, New Zealand, and Antarctic Birds. Vol. 3. Snipe to Pigeons*. Oxford University Press, Melbourne.

Hobson, R (2015). Squatter Pigeon Survey & Monitoring Proposal: Expedition National Park, Amphitheatre Section. Unpublished survey and monitoring project proposal to the Queensland National Parks and Wildlife Service.

Queensland Department of Environment and Heritage Protection (QLD DEHP) (2012). *Wildnet database records.*

Queensland Resources Council (2015). Submission on the proposed delisting of *Geophaps scripta scripta.*

Squatter Pigeon Workshop (2011). *Proceedings from the workshop for the Squatter Pigeon (southern). 14-15 December 2011*. Toowoomba Office of the Queensland Parks and Wildlife Service.

Stewart, D (2015). Personal communication by email, 10 April 2015. Queensland Department of Environment and Heritage Protection.

The Minister approved this conservation advice and included this species in the vulnerable category, effective from 04/07/2019.

Conservation Advice

Hirundapus caudacutus

White-throated Needletail

Taxonomy

Conventionally accepted as *Hirundapus caudacutus* Latham, 1801.

Other names: Needle-tailed, Spine-tailed or White-throated Swift, Needletail or Northern Needletail, Needle-tailed, Pin-tailed or Prickly Swallow, Prickly Tail or Prickly Swift, Storm Bird (Higgins 1999).

There are two recognised subspecies:

- subspecies *caudacutus* occurs in central and eastern Siberia, northern Mongolia, northern China and the Korean Peninsula, Sakhalin and Japan, and migrates to spend the non-breeding season in Australasia.
- subspecies *nudipes*, which breeds in the Himalayas from northern Pakistan to Assam and south-western China, and is largely resident and does not occur in Australasia (Chantler 1999; Higgins 1999).

Summary of assessment

Conservation status

Vulnerable: Criterion 1 A2(b)

The highest category for which *Hirundapus caudacutus* is eligible to be listed is Vulnerable.

Hirundapus caudacutus has been found to be eligible for listing under the following categories:
Criterion 1: A2(b): Vulnerable
Criterion 2: Not eligible
Criterion 3: Not eligible
Criterion 4: Not eligible
Criterion 5: Not eligible

The Victorian Scientific Advisory Committee undertook an assessment in 2016 and found White-throated Needletail eligible for listing. The White-throated Needletail is listed as threatened in Victoria under the *Flora and Fauna Guarantee Act 1988*.

For information on the listing status of this species under relevant state or territory legislation, see http://www.environment.gov.au/cgi-bin/sprat/public/sprat.pl

Reason for conservation assessment by the Threatened Species Scientific Committee

This advice follows assessment of information provided by a nomination from the public to list the White-throated Needletail.

Public consultation

Notice of the proposed amendment and a consultation document was made available for public comment for 37 business days between 31 October and 21 December 2018. Any comments received that were relevant to the survival of the species were considered by the Committee as part of the assessment process.

Species/sub-species information

Description

The White-throated Needletail is a large swift with a thickset, cigar-shaped body, stubby tail and long pointed wings (20 cm in length and approximately 115–120 g in weight). Sexes are alike, with no seasonal variation in plumage. The adults have a dark-olive head and neck, with an iridescent gloss on the crown; the mantle and the back are paler, greyish; and the upperwings are blackish, sometimes with a greenish gloss, with a contrasting white patch at the base of the trailing edge; the uppertail is black with a greenish gloss. The face is dark-olive with a narrow, white band across the forehead and lores and a white patch on the chin and throat. The underparts are generally dark-olive except for a U-shaped band across the rear flanks, the vent and the undertail coverts, and the undertail is black with a greenish gloss. The underwing is black brown with glossy grey-brown flight feathers. The bill is black, the eyes black-brown and the legs and feet are dark grey, sometimes with a pinkish tinge.

Juveniles have a similar appearance to the adults, but can be separated by duller plumage, with little gloss. The pale saddle is duller, contrasting less with the head, neck and uppertail; and the white band across the forehead and white patches on the upperwings and the vent and undertail coverts are all less prominent and duller (Higgins 1999).

The White-throated Needletail is generally gregarious when in Australia, sometimes occurring in large flocks, though they are occasionally seen singly. Occasionally the species occurs in mixed flocks with other aerial insectivores, including Fork-tailed Swifts (*Apus pacificus*) and Fairy Martins (*Hirundo ariel*) (Learmonth 1950, 1951; McMicking 1925; Wheeler 1959).

Distribution

The White-throated Needletail is widespread in eastern and south-eastern Australia (Barrett et al. 2003; Blakers et al. 1984; Higgins 1999). In eastern Australia, the species is recorded in all coastal regions of Queensland and NSW, extending inland to the western slopes of the Great Dividing Range and occasionally onto the adjacent inland plains. Further south on the mainland, it is widespread in Victoria, though more so on and south of the Great Dividing Range, and there are few records in western Victoria. The species occurs in adjacent areas of south-eastern South Australia, where it extends west to the Yorke Peninsula and the Mount Lofty Ranges. It is widespread in Tasmania (Barrett et al. 2003; Blakers et al. 1984; Higgins 1999).

White-throated Needletails only occur as vagrants in the Northern Territory (recorded in the Top End, including around Darwin, Katherine and Mataranka and Tennant Creek; and further south around Alice Springs) and in Western Australia (at disparate sites from the Mitchell Plateau in the Kimberley, south to the Nullarbor Plain and Augusta in the South West, and west to Barrow Island, the Houtman Abrolhos Islands and the Swan River Plain) (Barrett et al. 2003; Blakers et al. 1984; Brooker et al. 1979; Sedgwick 1978; Slater 1964; Storr 1987; Storr et al. 1986; Wheeler 1959). The species is also a vagrant to various outlying islands, including Norfolk, Lord Howe, Macquarie, Christmas and Cocos-Keeling Islands (Barrand 2005; Green 1989; McAllan et al. 2004; Schodde et al. 1983; Stokes et al. 1984; Warham 1961).

The breeding distribution of the White-throated Needletail is fragmented, with two subspecies occurring in different parts of Asia. The nominate subspecies *H. c. caudacutus* breeds from northern Japan west to central and eastern Siberia, while subspecies *H. c. nudipes* breeds from south-western China to northern Pakistan, and is largely resident (Chantler 1999).

Relevant biology/ecology

General habitat

In Australia, the White-throated Needletail is mostly aerial, from heights of less than 1 m up to more than 1000 m above the ground (Coventry 1989; Tarburton 1993). Although they occur over most types of habitat, they are recorded most often above wooded areas, including open forest and rainforest, and may also fly below the canopy between trees or in clearings (Higgins 1999). When flying above farmland, they are more often recorded above partly cleared pasture, plantations or remnant vegetation at the edge of paddocks (Emison & Porter 1978; Friend 1982; Tarburton 1993). In coastal areas, they have been observed flying over sandy beaches or mudflats (Cooper 1971; Crompton 1936; Davis 1965), and often around coastal cliffs and other areas with prominent updraughts, such as ridges and sand-dunes (Cooper 1971; Dawson et al. 1991; Loyn 1980; Mitchell et al. 1996; Schulz & Kristensen 1994).

Roosting habitat

The species roosts in trees amongst dense foliage in the canopy or in hollows (Corben et al. 1982; Day 1993; Quested 1982; Tarburton 1993, 2015).

Feeding

During the non-breeding season in Australia, the White-throated Needletail has been recorded eating a wide variety of insects, including beetles, cicadas, flying ants, bees, wasps, flies, termites, moths, locusts and grasshoppers (Cameron 1968; Madden 1982; Rose 1997; Tarburton 1993).

Life history

The species does not breed in Australia (Higgins 1999). The White-throated Needletail lays eggs from late May to early June in their breeding grounds in the Northern Hemisphere (Chantler 1999). The nest is placed in a vertical hollow in a tall coniferous tree or on a vertical rock-face, either comprising a small bracket or half-cup of thin twigs and straw cemented together by the bird's saliva and glued to the side of the hollow or rock (Roberts 1991), or a shallow scrape among debris accumulated at the bottom of a tree hollow (Chantler 1999). Clutches usually comprise two eggs (Dement'ev & Gladkov 1951; Yamashina 1962) but some may be as large as seven eggs (Chantler 1999), and these are incubated by both sexes for 40 days (Chantler 1999). The chicks fledge after 40–42 days (Chantler 1999; Dement'ev & Gladkov 1951; Yamashina 1962).

There are no published details of the ages of sexual maturity or life expectancy of the White-throated Needletail, however, the estimated generation time is 8.5 years (BirdLife International 2018).

Movement patterns

The nominate subspecies *caudacutus* is a trans-equatorial migrant, breeding in the Northern Hemisphere and flying south for the boreal winter (Higgins 1999).

Departure from breeding grounds

The species breeds in wooded lowlands and sparsely vegetated hills, as well as mountains covered with coniferous forests in eastern Siberia, north-eastern China, the Korean Peninsula and Japan. The species leaves the breeding grounds between late August and October, flying singly or in scattered flocks (Chantler 1999; Dement'ev & Gladkov 1951).

The southern passage from the breeding grounds takes needletails through eastern China and Japan between August and November (Dement'ev & Gladkov 1951), and the Korean Peninsula mainly between September and October (Gore & Won 1971). Between late September and late November, most birds apparently migrate through Borneo and along the Malay Peninsula (Higgins 1999; M. Tarburton pers. Comm.). Passage may be extremely rapid and thus poorly detected (White & Bruce 1986). In Papua New Guinea, most records, presumably of birds on southern passage, occur between September and November (Bell 1970; Coates 1985; Hicks 1990; Rand & Gilliard 1967).

Non-breeding season in Australia

White-throated Needletails mainly enter Australia via the Torres Strait, usually during September and October, and sometimes in early November (Draffan et al. 1983; Warham 1962), and less often via the Arafura Sea (Warham 1962). The mean date of the first sighting in Australia is 22 October ± 27.62 days (range of 1 September and 27 December (Higgins 1999)). After reaching Australia, they move south along both sides of the Great Dividing Range in Queensland and NSW in October and November, usually arriving in southern parts of their range (Victoria and Tasmania) in November, with increasing numbers recorded from December and peaking in March (Emison et al. 1987; Higgins 1999).

Northern passage

Northward migration from Australia begins between mid-March and April (Higgins 1999). A few birds occasionally remain in Australia during the breeding season (Higgins 1999).

When undertaking northern migration to return to their breeding grounds in the Northern Hemisphere, the majority of the White-throated Needletail population pass through New Guinea in March and April (Eastwood & Gregory 1995; Hicks 1990) and are thought to mostly travel east of Borneo (Smythies 1957, 1981). There are records of birds on northward passage through Indonesia in March and April (Coates & Bishop 1997; Smythies 1957, 1981; White & Bruce 1986), and there are records from the Malay Peninsula, between March and mid-May (Medway & Wells 1976; Wells 1999). They are also recorded passing through Hong Kong between mid-March and mid-May (Chalmers 1986; Chantler & Driessens 1995), and eastern China in May.

White-throated Needletails arrive back at their breeding grounds in the Northern Hemisphere in mid-May (Chantler 1999; Chantler & Driessens 1995; Dement'ev & Gladkov 1951).

Threats

In Australia there is evidence of collision with wind turbines (Hull 2013), overhead wires (Cameron & Hinchey 1981; Campbell 1930; Wheeler 1965), windows (Slater 1964) and lighthouses (Draffan et al. 1983; Stokes 1983) but the scale of impact at the population level requires further investigation.

Tarburton (2014) identified the use of insecticides, particularly organochlorines, as another possible cause of decline of White-throated Needletails, either through a decrease in the abundance of invertebrates from wide use of insecticides or from secondary poisoning by insecticides accumulated as sublethal doses in the prey.

As noted in Tarburton (2014), the loss of roosting sites in Australia may also be contributing to the decline of the species. Loss of forest and woodland habitats may have also resulted in the reduction of invertebrate prey.

It is thought that logging of taiga forests in Siberia, where most of the population breeds, poses the greatest risk by removing old trees and stumps that contain hollows which this species uses to breed (Newell et al. 2000; Crowley 2005; Smirnov et al. 2013).

On the species' breeding grounds it was formerly hunted with nets placed near their breeding sites.

Table 1: Threats impacting the White-throated Needletail in approximate order of severity of risk, based on available evidence

Number	Threat factor	Threat type and status	Evidence base
1.0	Habitat loss and fragmentation		
1.1	Logging of breeding habitat	suspected current	The loss of old, hollow bearing trees in the breeding range in the northern hemisphere is suspected be impacting breeding success (Tarburton 2014).
1.2	Loss of habitat in the non-breeding range	suspected current	The loss of roosting sites in Australia may be contributing to the decline of the species. Loss of forest and woodland habitats may have also resulted in the reduction of invertebrate prey (Tarburton 2014).
2.0	Direct mortality		
2.1	Wind turbines and overhead wires	known current	Impacts from wind farms can be categorised as direct (collisions with wind turbines) and indirect (barrier and alienation, with the potential to reduce access to habitat). Collision with wind turbines and overhead wires is of low severity and affects a small number of birds (Hull 2013)
3.0	Poisoning		
3.1	Organochlorines	potential	Tarburton (2014) identified the use of insecticides, particularly organochlorines, as a possible cause of decline of White-throated Needletail, either through a decrease in the abundance of invertebrates from wide use of insecticides or from secondary poisoning by insecticides accumulated as sublethal doses in the prey.

How judged by the Committee in relation to the EPBC Act criteria and regulations

Criterion 1. Population size reduction (reduction in total numbers) Population reduction (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

Evidence:

Eligible under Criterion 1 A2(b) for listing as Vulnerable

Tarburton (2014) reported that based on data collected between 1998 and 2002, the *New Atlas of Australian Birds* (Barrett et al. 2003) indicated a 49 per cent decline in reporting rates (number of records as a proportion of number of surveys, adjusted for the survey method, season and size of area searched) of needletails compared with those of the first *Atlas of Australian Birds* conducted between 1977 and 1981 (Blakers et al. 1984).

Tarburton (2014) showed that with each decade after 1950 a progressive decline in the mean number of needletails counted per flock has occurred. Australia-wide trends in mean number of White-throated Needletails counted per flock have fallen from 164 ± 37.3 in 1951-1960 to 42 ± 1.7 in 2001-2010 (Tarburton 2014). These declines are continuing with more recent data indicating that the mean number of White-throated Needletails counted per flock between 2011-2017 has fallen to 36 ± 0.9.

Tarburton (2014) demonstrated that from three sites in Victoria, at the level of each eastern state and at the national scale, a 30-50 per cent decline in White-throated Needletail flock size has occurred over three generations (25.5 years).

The Committee considers that the species has undergone a substantial reduction in numbers over three generation lengths (25.5 years for this assessment), equivalent to at least 30 – 50 percent and the reduction has not ceased, the cause has not ceased and is not understood. Therefore, the species 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 subpopulations; (v) number of mature individuals			
(c) Extreme fluctuations in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) number of locations or subpopulations; (iv) number of mature individuals			

Evidence:

Not eligible

Within Australia, the extent of occurrence is estimated at >20,000 sq km, and the area of occupancy estimated at >18,000 sq km. These figures are based on the mapping of point records from post 1997 species observations, obtained from state governments, museums, CSIRO, and Birdlife Australia. 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 (DotE 2015). Therefore, the species has not met a required element of this criterion.

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 subpopulation	**≤ 50**	**≤ 250**	**≤ 1,000**
(a) (ii) % of mature individuals in one subpopulation =	**90 – 100%**	**95 – 100%**	**100%**
(b) Extreme fluctuations in the number of mature individuals			

Evidence:

Not eligible

Within Australia, the population size has not been quantified, but it is not believed to approach the thresholds for Vulnerable under the population size criterion (<10,000 mature individuals with a continuing decline estimated to be >10 per cent in ten years or three generations, or with a specified population structure) (BirdLife International 2018). Therefore, the species has not met this required element of this criterion.

Criterion 4. Number of mature individuals			
	Critically Endangered Extremely low	**Endangered Very Low**	**Vulnerable Low**
Number of mature individuals	**< 50**	**< 250**	**< 1,000**

Evidence:

Not eligible

The global population size has not been quantified, but the species is reported to be local and uncommon throughout much of its range (del Hoyo *et al*. 1999). Within Australia, the population size has not been quantified (BirdLife International 2018), but it is not believed to approach the thresholds for Vulnerable under the population size criterion. Therefore, the species has not met this required element of this criterion.

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**

Evidence:

Not eligible

Population viability analysis has not been undertaken.

Conservation actions

Recovery plan

A Recovery Plan is not required; an approved Conservation Advice for the species provides sufficient direction to implement priority actions, mitigate against key threats and enable recovery. Management and research activities are being undertaken at international, national, state and local levels.

Primary conservation actions

Work with governments in East Asia to minimise destruction of key breeding habitats.

Important habitats in Australia are identified and protected.

Conservation and Management priorities

- Habitat loss and modifications
  - Seek the support of governments in East Asia to protect remaining old growth forests within the breeding range of the species.
  - Identify requirements of important habitat in Australia.
  - Support initiatives to improve habitat management at key sites in Australia.

Stakeholder Engagement

- Through the bilateral migratory bird consultative meetings with the Governments of Japan, China and the Republic of Korea, raise awareness of the conservation of White-throated Needletail.
- Promote the conservation, and raise the profile, of White-throated Needletail through strategic programs and educational products.
- Promote the exchange of information between governments, NGOs and communities through use of networks, publications and websites.

Survey and Monitoring priorities

- Enhance existing White-throated Needletail monitoring programs, such as BirdLife Australia's *Swift Monitoring Sites*, particularly to improve coverage in under surveyed parts of Australia.

Information and Research priorities

- Use remote sensing to assess the extent of habitat loss at the breeding grounds.
- Undertake work to more precisely assess White-throated Needletail life history, population size, distribution and ecological requirements in Australia.
- Improve knowledge about potential threatening processes including the impacts of infrastructure (i.e. wind turbines and overhead wires).
- Quantify levels of organochlorines in individuals and prey species.

Recommendations

(i) The Committee recommends that the list referred to in section 178 of the EPBC Act be amended by **including** in the list in the Vulnerable category:

Hirundapus caudacutus

(ii) The Committee recommends that there not be a recovery plan for this species.

Threatened Species Scientific Committee

27/02/2019

References cited in the advice

Barrand, P.D. (2005). White-throated Needletail *Hirundapus caudacutus*: a new bird for Christmas Island. *Australian Field Ornithology*. 22:104-105.

Barrett, G., A. Silcocks, S. Barry, R. Cunningham & R. Poulter (2003). *The New Atlas of Australian Birds*. Melbourne, Victoria: Birds Australia.

Bell, H.L. (1970). Field notes on birds of the Nomad River sub-district, Papua. *Emu*. 70:97-104.

BirdLife International (2018) Species factsheet: *Hirundapus caudacutus*. Downloaded from http://www.birdlife.org on 06/08/2018.

Blakers, M., S.J.J.F. Davies & P.N. Reilly (1984). *The Atlas of Australian Birds*. Melbourne, Victoria: Melbourne University Press.

Brazil, M. (2009). *Birds of East Asia: eastern China, Taiwan, Korea, Japan, eastern Russia*. Christopher Helm, London.

Brooker, M.G., M.G. Ridpath, A.J. Estbergs, J. Bywater, D.S. Hart & M.S. Jones (1979). Bird observations on the north-western Nullarbor Plain and neighbouring regions, 1967-1978. *Emu*. 79:176-190.

Cameron, A.C. (1968). Feeding habits of the Spine-tailed Swift. *Emu*. 68:217-219.

Cameron, R. & M. Hinchey (1981). An apparently immature White-throated Needletail in Australia. *Australian Bird Watcher*. 9:68.

Campbell, A.G. (1930). Notes on swifts. *Emu*. 29:308.

Chalmers, M.L. (1986). *Birds of Hong Kong*. Hong Kong: Hong Kong Bird Watching Society.

Chantler, P. (1999). Apodidae (swifts) species accounts. **In:** del Hoyo, J., A. Elliott & J. Sargatal, eds. *Handbook of the Birds of the World. Volume 5. Barn-owls to Hummingbirds*. Page(s) 419-457. Barcelona: Lynx Editions.

Chantler, P. & G. Driessens (1995). *Swifts*. Robertsbridge, UK: Pica Press.

Coates, B.J. (1985). *The Birds of Papua New Guinea. Volume 1*. Alderley, Queensland: Dove Publications.

Coates, B.J. & K.D. Bishop (1997). *A Guide to the Birds of Wallacea Sulawesi, The Moluccas and Lesser Sunda Islands, Indonesia*. Alderley, Queensland: Dove Publications.

Cooper, R.P. (1971). High flying swifts. *Australian Bird Watcher*. 4:79-80.

Corben, C., G. Roberts & A. Smyth (1982). Roosting of a White-throated Needletail. *Sunbird*. 12:47-48.

Coventry, P. (1989). Comments on airborne sightings of White-throated Needletails *Hirundapus caudacutus*. *Australian Bird Watcher*. 13:36-37.

Crompton, A. (1936). Spine-tailed Swift *Hirundapus caudacutus*. *South Australian Ornithologist*. 13:183-184.

Crowley, R.M. (2005). Stepping onto a moving train: The collision of illegal logging, forestry policy and emerging free trade in the Russian Far East. *Pacific Rim Law and Policy Journal* 14: 425-454.

Davis, W.A. (1965). Field notes from South Gippsland. *Australian Bird Watcher*. 2:134-138.

Dawson, P., D. Dawson, I. Reynolds & S. Reynolds (1991). Notes on the birds of Logan Reserve, southeast Queensland, 1967-1990. *Sunbird*. 21:93--111.

Day, N. (1993). Tree perching and presumed roosting of White-throated Needletails *Hirundapus caudacutus*. *Australian Bird Watcher*. 15:43-44.

del Hoyo, J., Elliott, A. & Sargatal, J. (1999). Handbook of the Birds of the World. Lynx Editions, Barcelona.

Dement'ev, G.P. & N.A. Gladkov (1951). *Birds of the Soviet Union. Volume 1*. Jerusalem: Israel Program for Scientific Translations (1969).

Draffan, R.D.W., S.T. Garnett & G.J. Malone (1983). Birds of the Torres Strait: an annotated list and biogeographic analysis. *Emu*. 83:207-234.

Eastwood, C. & P. Gregory (1995). Interesting sightings during 1993 & 1994. *Muruk*. 7:128-142.

Emison, W.B. & J.W. Porter (1978). Summer surveys of birds in the Mt Cobberas - Snowy River area of Victoria, Australia. *Emu*. 78:126-136.

Emison,W.B., C.M. Beardsell, F.I. Norman, R.H. Loyn & S.C. Bennett (1987). *Atlas of Victorian Birds*. Melbourne: Department of Conservation (Forest & Lands) & Royal Australian Ornithological Union.

Friend, G.R. (1982). Bird populations in exotic pine plantations and indigenous eucalypt forests in Gippsland, Victoria. *Emu*. 82:80-91.

Gore, M.E.J. & P.-O. Won (1971). *The Birds of Korea*. Seoul, Korea: Taewon Publishing

Green, R.H. (1989). *Birds of Tasmania*. Launceston, Tasmania: Potoroo Publishing.

Hicks, R.K. (1990). Arrival and departure dates in the Port Moresby area of migrants from the north. *Muruk*. 4:91-105.

Higgins, P.J. (ed.) (1999). *Handbook of Australian, New Zealand and Antarctic Birds. Volume Four - Parrots to Dollarbird*. Melbourne: Oxford University Press.

Hull, C.L., Stark, E.M., Peruzzo, S. & Sims, C.C. (2013). Avian collisions at two wind farms in Tasmania, Australia: taxonomic and ecological characteristics of colliders versus noncolliders. *New Zealand Journal of Zoology* 40, 47-62.

Learmonth, N.F. (1950). Observations on swifts near Portland, Vic., during summer, 1949-1950. *Emu*. 50:56-58.

Learmonth, N.F. (1951). More observations on swifts. *Emu*. 51:151-152.

Loyn, R.H. (1980). Bird populations in a mixed eucalypt forest used for production of wood in Gippsland, Victoria. *Emu*. 80:145-156.

Madden, J.L. (1982). Avian predation of the Woodwasp, *Sirex noctilio* F., and its parasitoid complex in Tasmania. *Australian Wildlife Research*. 9:135-144.

McAllan, I.A.W., B.R. Curtis, I. Hutton & R.M. Cooper (2004). The birds of the Lord Howe Island Group: a review of records. *Australian Field Ornithology*. 21:1-82.

McMicking, F.V. (1925). Spine-tailed Swifts *(Hirundapus caudacutus)* feeding on grasshoppers. *Emu*. 25:41.

Medway, L. & D.R. Wells (1976). *The Birds of the Malay Peninsula, Volume 5*. Witherby, London.

Mitchell, A., J. Peter & G. McCarthy (1996). Birds of the Ironbark Basin. *Geelong Bird Report*. 1995:29-40.

Newell, J., Lebedev, A., Gordon, D. & Rees, M. (2000). Plundering Russia's Far Eastern Taiga: Illegal Logging, Corruption and Trade. An Investigation into the Extent of Illegal Logging and Trade in the Region, along with Recommendations. Friends of the Earth Japan, Tokyo.

Pescott, T. (1983). *Birds of Geelong*. Geelong: Neptune Press.

Quested, T. (1982). Spine-tailed Swift landing in tree. *Australian Birds*. 16:64.

Rand, A.L. & E.T. Gilliard (1967). *Handbook of New Guinea Birds*. London: Weidenfeld & Nicolson.

Roberts, T.J. (1991). *The Birds of Pakistan. Volume 1, Non-passeriformes*. Karachi, Pakistan: Oxford University Press.

Rose, A.B. (1997). Notes on the diet of swifts, kingfishers and allies in eastern New South Wales. *Australian Bird Watcher*. 17:203-210.

Schodde, R., P. Fullagar & N. Hermes (1983). A review of Norfolk Island birds: past and present. *Australian National Parks and Wildlife Service Special Publication*. 8.

Schulz, M. & K. Kristensen (1994). Notes on selected bird species on the south-western coast of Tasmania, between Port Davey and Cape Sorell. *Australian Bird Watcher*. 15:265-272.

Sedgwick, E.H. (1978). A population study of Barrow Island avifauna. *West Australian Naturalist*. 14:85-108.

Slater, K.R. (1964). Spine-tailed Swift in central Australia. *Emu*. 64:72.

Smirnov, D.Y., Kabanets, A.G., Milakovsky, B.J., Lepeshkin, E.A. & Sychikov, D.V. (eds) (2013). Illegal Logging in the Russian Far East: Global Demand and Taiga Destruction. World Wildlife Fund, Moscow.

Smythies, B.E. (1957). *Chaetura caudacuta* in Borneo. *Ibis*. 99:687-688.

Smythies, B.E. (1981). *The Birds of Borneo*. Sabah, Kuala Lumpur: Sabah Society/Malayan Nature Society.

Stokes, T. (1983). Bird casualties in 1975-76 at the Booby Island Lightstation, Torres Strait. *Sunbird*. 13:53-58.

Stokes, T., W. Sheils & K. Dunn (1984). Birds of the Cocos - Keeling Islands, Indian Ocean. *Emu*. 84:23-28.

Storr, G.M. (1987). Birds of the Eucla Division of Western Australia. *Records of the Western Australian Museum*. Suppl. 27.

Storr, G.M., R.E. Johnstone & P. Griffin (1986). Birds of the Houtman Abrolhos, Western Australia. *Records of the Western Australian Museum Supplement*. 24.

Tarburton, M.K. (1993). Radiotracking a White-throated Needletail to roost. *Emu*. 93:121--124.

Tarburton, M.K. (2014). Status of the White-throated Needletail *Hirundapus caudacutus* in Australia: Evidence for a marked decline. *Australian Field Ornithology* 31, 122-140.

Tarburton. M. (2015) Swifts over the Hunter Region. *The Whistler* 9: 23-27.

Warham, J. (1961). A Spine-tailed Swift at Macquarie Island. *Emu*. 61:189-190.

Warham, J. (1962). Bird islands within the Barrier Reef and Torres Strait. *Emu*. 62:99-111.

Wells, D.R. (1999). *The Birds of the Malay Peninsula*. San Diego: Academic Press.

Wheeler, J. (1965). A boy's care of an injured bird. *Australian Bird Watcher*. 2:152-153.

Wheeler, W.R. (1959). Notes on Swifts, 1958-59. *Bird Observer*. 334:2-5.

White, C.M.N. & M.D. Bruce (1986). The birds of Wallacea. *B.O.U. Check-list*. 7.

Yamashina, Y. (1962). *Birds of Japan - A Field Guide*. Tokyo: Tokyo News Service.