Threatened Alpine and Subalpine Ecosystems of Australia

An IUCN Red List of Ecosystems Assessment Version 0.5

Lead authors

Jessica A. Rowland, Chloe Sato, Tracey Regan, David A. Keith, José-R. Ferrer-Paris, Rebecca Lester, Ella Plumanns Pouton, Arn Tolsma, Michael Traurig, Clare Vernon, Susanna Venn, Emily Nicholson

IUCN
RED LIST OF ECOSYSTEMS
THE UNIVERSITY OF MELBOURNE
UNSW SYDNEY
VICTORIA State Government | Energy, Environment and Climate Action
Australian Government
Australian Research Council
DEAKIN UNIVERSITY

Authors and Contributors

Lead authors/editors

Jessica A. Rowland	Deakin University, Australia Monash University, Australia IUCN Commission on Ecosystem Management
Chloe Sato	Parks Australia
Tracey J. Regan	Arthur Rylah Institute for Environmental Research, Department of Energy Environment and Climate Action (DEECA)
David A. Keith	University of New South Wales, Australia IUCN Commission on Ecosystem Management
José-R. Ferrer-Paris	University of New South Wales, Australia
Rebecca Lester	Deakin University, Australia
Ella Plumanns Pouton	Deakin University, Australia
Arn Tolsma	Arthur Rylah Institute for Environmental Research, Department of Energy Environment and Climate Action (DEECA)
Michael Traurig	Deakin University, Australia
Clare Vernon	Deakin University, Australia
Susanna Venn	Deakin University, Australia
Emily Nicholson	University of Melbourne, Australia IUCN Commission on Ecosystem Management

Contributors

Jane Balmer	Department of Natural Resources and Environment Tasmania, Australia
Leon Barmuta	University of Tasmania, Australia
James Camac	University of Melbourne, Australia
Michael Driessen	University of Tasmania, Australia
Casey Gibson	University of New South Wales, Australia
Ken Green	Australian National University, Australia
Calvin Lee	ERM, Hong Kong
Mark Lintermans	University of Canberra
Jamie Kirkpatrick	University of Tasmania, Australia
Keith McDougall	La Trobe University, Australia
John Morgan	La Trobe University, Australia
Annette Muir	Arthur Rylah Institute for Environmental Research, Department of Energy Environment and Climate Action (DEECA)
Nicholas Murray	James Cook University, Australia
Tarmo Raadik	Arthur Rylah Institute for Environmental Research, Department of Energy Environment and Climate Action (DEECA)
Emma Sumner	Western Sydney University, Australia
Phil Suter	La Trobe University, Australia
Anikó Tóth	University of New South Wales, Australia
Kate Umbers	Western Sydney University, Australia
Brodie Verrall	Queensland Herbarium, Australia
Jessica Walsh	Monash University, Australia
Jennie Whinam	University of Tasmania, Australia
Matt White	Arthur Rylah Institute for Environmental Research, Department of Energy Environment and Climate Action (DEECA)

Genevieve Wright	Department of Climate Change, Energy, the Environment and Water (DEECA), Australia
Dick Williams	La Trobe University, Australia

Citation: Rowland JA et al. 2025. Threatened Alpine and Subalpine Ecosystems of Australia: An IUCN Red List of Ecosystems Assessment.

DOI: TBD
ISBN: TBD
Cover Photos: Clockwise from top left: © Susanna Venn, Clare Vernon, Jamie Kirkpatrick, Grant Dixon

Acknowledgements

The Alpine and Subalpine Ecosystems of Australia are on the lands of the Taungurung, Gunai Kurnai, Walgalu, Wiradjuri, Ngarigo, Ngunawal, Gundungurra, and the Palawa. We acknowledge their longstanding connection and care for the Alpine Country.

This work was made possible by funding from The Australian Research Council (LP170101143).

Photo: Susanna Venn.

Table of contents

Acronyms

Acronym	Meaning
ACT	Australian Capital Territory
AOO	Area of Occupancy
ASL	Above sea level
EVC	Ecological Vegetation Classes
EOO	Extent of Occurrence
MVG	Major Vegetation Group
MVS	Major Vegetation Subgroup
NSW	New South Wales
NVIS	National Vegetation Information System
IUCN	International Union for Conservation of Nature

Executive summary

Australian alpine and subalpine ecosystems support unique, diverse biota and provide important ecosystem services such as fresh water. They exist on geographical and climate extremities, making them especially vulnerable to climate change. The region has a history of livestock grazing, invasive hard-hooved herbivores, infrastructure construction (e.g., ski resorts, road infrastructure, power lines, and dams), and altered fire regimes. These multiple interacting drivers may be putting Australian Alpine ecosystems at risk. Understanding the risks to these ecosystem types and the underlying causes is essential to planning appropriate and timely conservation, developing on-ground monitoring programs, and setting policies and regulations for land use.

In this report, we document the IUCN Red List of Ecosystems assessment for Australian alpine and subalpine ecosystem types. This project brought together leading risk assessment experts and alpine ecosystem ecologists from across Australia to:

- Develop a national typology of alpine and subalpine ecosystem types;
- Create a national map of alpine and subalpine ecosystem types; and
- Undertake IUCN Red List of Ecosystems assessments for 15 ecosystem types, including developing conceptual model, ecosystem description, identifying key threatening processes, compiling existing datasets, and assessing changes in the distribution, environmental conditions, and biota each ecosystem type.

Outputs from these assessments are intended to be used to support conservation decision making in Australian Alpine ecosystems.

Methods

We used the IUCN Red List of Ecosystems version 1.1 to assess the threat status of the 15 Australian alpine and subalpine ecosystem types. IUCN Red List of Ecosystems is the global standard for assessing the risk of ecosystem collapse of the world's ecosystems. The framework assesses collapse risk based on five quantitative criteria that capture spatial and functional symptoms of decline.

To complete these assessments, we held a 4-day in-person workshop followed by a half-day virtual workshop with a team of experts in alpine ecosystems and IUCN Red List of Ecosystems. The experts worked within research, policy, and land management in university research groups, and/or state and national government departments and organisations across all relevant jurisdictions. Each assessment was completed by a small group consisting of experts in the ecosystem type and an IUCN Red List of Ecosystems expert. Each assessed was reviewed by at least two experts not involved in the specific assessment.

Outcomes

Seven (47%) of the 15 ecosystem types were classified as threatened, including one Endangered and five Vulnerable ecosystem types. This equates to 553.47 km^2 (7.25% of the study area) covered by threatened ecosystem types. Three ecosystem types were assessed as Near Threatened, and three as Least Concern. One ecosystem type, *Alpine-subalpine Fen,* was assessed as Data Deficient. In total, 67% ecosystem types are threatened or Near Threatened, covering 83% of the alpine and subalpine region.

Most ecosystem types were listed as threatened or Near Threatened due to declines in distribution or having a restricted distribution and ongoing threats, including future declines in environmental suitability under climate change, altered fire regimes, and hard-hooved animals. However, two ecosystem types were listed as threatened due to declines in integrity: *Snowpatch Herbfield* due to snowmelt date and *Alpine-subalpine Lakes* due to changes in microinvertebrate assemblage condition

Analysis showed that for several ecosystem types, fire regimes are not predicted to increase to a frequency likely to cause large declines in the next 50 years. However, fire frequency has increased in recent years in *Alpine-subalpine Woodland,* contributing towards listing as Near Threatened.

We were only able to assess Criteria E for *Alpine-subalpine Streams.* The ecosystem type was listed as critically Endangered due to modelled future declines in water flow.

Sufficient data were not available to assess collapse risk due to changes in the biotic features and processes for many ecosystem types, across Criteria C and D, a key limiting factor in producing reliable assessments of future risk.

Dominant threats (most commonly list as a threat among ecosystem types) were climate change-related threatening processes, including temperature extremes, drought, and changes to fire regimes. trampling, herbivory and wallowing by invasive ungulate species, and human land uses, including infrastructure, recreational activities, and farming were also common threats among many ecosystem types. We summarise assessment outcomes for each ecosystem type in Table 1.

Table 1. Relationship between the ecosystem units in this project, the IUCN global ecosystem typology (Keith et al. 2022a), typology of alpine and subalpine ecosystems of Australia (Venn et al. 2017), and the IUCN RLE Assessment outcomes.

Ecosystem Units	**IUCN Global Ecosystem Typology**	**IUCN RLE Assessment (**values in parentheses show plausible bounds)
Tasmanian Palaeoendemic Forest and Woodland	T2.3 Oceanic temperate rainforests	**Vulnerable** (Near Threatened – Endangered)
Alpine-subalpine Woodland and Forest	T4.4 Temperate woodlands	**Near Threatened**
Alpine-subalpine Open Grassy Heathland and Herbfield	T6.4 Temperate alpine meadows and shrublands	**Least Concern** (Least Concern – Near Threatened)
Alpine-subalpine Closed Heath	T6.4 Temperate alpine meadows and shrublands	**Least Concern**
Coniferous Heath	T6.4 Temperate alpine meadows and shrublands	**Vulnerable** (Vulnerable – Endangered)
Fjaeldmark/feldmark	T6.4 Temperate alpine meadows and shrublands	**Vulnerable**
Snowpatch Herbfield	T6.4 Temperate alpine meadows and shrublands	**Endangered** (Vulnerable – Endangered)
Cushion Moorland	T6.4 Temperate alpine meadows and shrublands	**Vulnerable**
Tasmanian Alpine Heath	T6.4 Temperate alpine meadows and shrublands	**Near Threatened** (Least Concern – Near Threatened)
Tasmanian Alpine Sedgeland	T6.4 Temperate alpine meadows and shrublands	**Near Threatened** (Least Concern – Near Threatened)
Alpine-subalpine Damp Valley Grassland and Rushland	T6.4 Temperate alpine meadows and shrublands	**Least Concern**
Alpine *Sphagnum* Bog and Associated Fen (nationally)	TF1.5 Boreal, temperate and montane peat bogs	**Least Concern** (Least Concern – Near Threatened)
Alpine *Sphagnum* Bog and Associated Fen (mainland)	TF1.5 Boreal, temperate and montane peat bogs	**Vulnerable** (Vulnerable – Endangered)
Alpine *Sphagnum* Bog and Associated Fen (Tasmania)	TF1.5 Boreal, temperate and montane peat bogs	**Endangered** (Endangered – Endangered)
Alpine-subalpine Fen	TF1.5 Boreal, temperate and montane peat bogs	**Data Deficient**
Alpine-subalpine Streams	F1.3 Freeze-thaw rivers and streams	**Endangered**
Alpine-subalpine Lakes	F2.4 Freeze-thaw freshwater lakes	**Critically Endangered**

Conclusion

This IUCN Red List of Ecosystems assessment has demonstrated there is a variable risk of collapsing among Australian alpine and subalpine ecosystem types. While many are not imperilled, others are at higher risk, particularly those that are already patchy and rely on narrow environmental conditions to persist. Climate change is a critical threat in the region, yet this is challenging to directly manage at a local or regional level. Effective management of on-ground threats such as invasive species and human activities will likely be essential to improve the integrity of these ecosystems and thus reduce their vulnerability to climate change related threats.

Understanding the threat status of these ecosystem types and the primary drivers of decline is important information to strengthen capacity to effectively monitor and conserve these unique ecosystems. For example, these assessments have already been used to inform threat management priority setting to better conserve *Alpine Sphagnum Bog and Associated Fen* under climate change (Regan et al. 2020).

This project provides a first attempt at collating the available spatial information to create a national map and typology of alpine and subalpine ecosystem types. However, the reliability of this output was limited by the low quality and inconsistencies in the state-level ecosystem maps. The development of a national typology was also constrained by the requirement to use existing classifications under the Environment Protection and Biodiversity Conservation Act 1999 (e.g., *Alpine Sphagnum Bog and Associated Fen*). Revising the classification approach is advised in future reassessments of these ecosystem types, especially the water-dependent systems.

There were insufficient data to assess many of the indicators of ecosystem integrity that were identified as important for evaluating the risk of collapse. This resulted in many ecosystem types only being assessed using the spatial symptoms of collapse or using only one or two indicators of integrity, increasing the uncertainty in the risk outcome. Capturing changes in environmental conditions and the characteristic biota are integral to reliably estimating collapse risk, especially in future predictions of risk. For the freshwater ecosystems, lakes and streams, there was insufficient data to assess any criteria. This highlights the importance of setting up ongoing monitoring efforts to understand ecological integrity and to understand progress towards degradation, based on relevant indicators for each ecosystem type. Two ecosystems, *Cushion Moorland* and *Tasmanian Alpine Sedgeland*, were also deficit of fundamental data on the relationships between environmental drivers and biotic change and what level of change constituted a threat. This made it difficult to determine the thresholds of collapse and relative severity for indicators, and thus quantify risk of degradation. This paucity of data demonstrates the value in further fundamental empirical work to quantify biotic responses to environmental change.

Future reassessments would benefit from additional analyses to enable reliable predictions of future risk under climate change. In particular, expanding the assessment of future fire risk to Tasmanian ecosystems, as FROST or other stochastic fire regime simulators become available for the region. It would also be valuable to assess likely future change in moisture balance in the full suite of ecosystem types, and to be able to estimate shifts in the distribution of characteristic species in response to changes, temperature, and precipitation.

Collaboration among experts in the IUCN Red List of Ecosystems, alpine ecosystem ecology, and ecological and climate modellers was important for providing and improving the evidence to support these assessments. This demonstrates the value in forming diverse disciplinary teams from across government, academia, and non-for-profit to effectively assess risk to ecosystems.

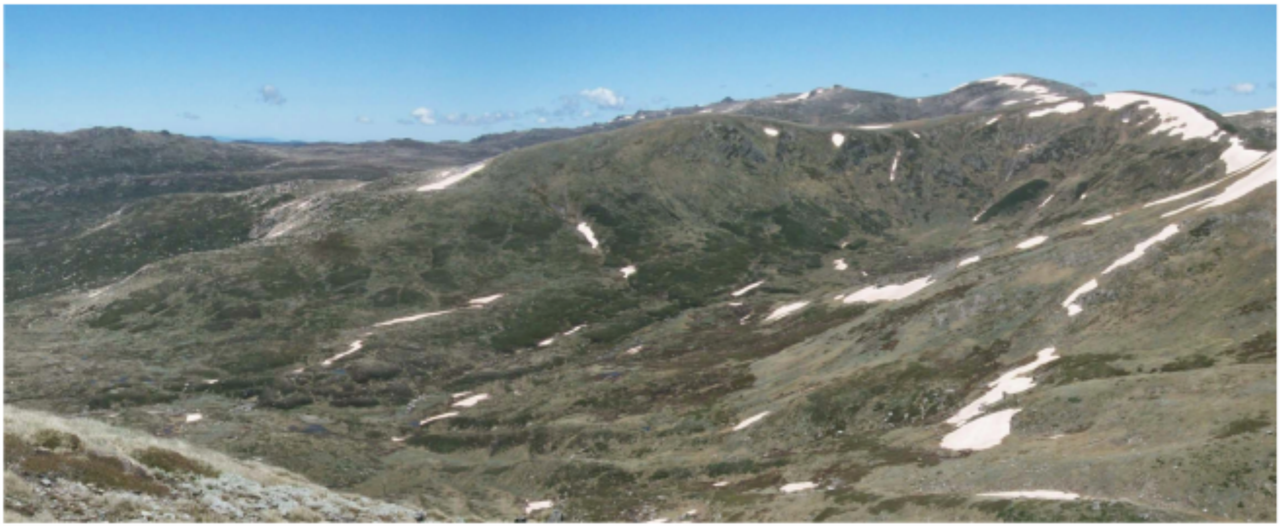


Photo: Susanna Venn.

Introduction

Australian alpine and subalpine region

The alpine and subalpine region of Australian extends across the Australian Alps on the mainland and the Central Plateau in Tasmania. The region includes terrestrial and freshwater ecosystems that cover 7624.21km^2 (0.09% of the country). The region is found > 600 m above sea level in Tasmania, > 1000 m on the mainland. In this report, the alpine ecosystems are defined as those occurring above the tree line (i.e., zone where tree growth becomes limited due to temperature; Venn et al. 2017), and the subalpine ecosystems are those that occur in mountains below the tree line. The Australian alpine region has a relatively low elevation, with the highest point being Mount Kosciuszko at 2228 m. The climate is cool and often has strong winds, with mean annual temperature below 8°C and no permanent snowline (Venn et al. 2017).

Australian alpine and subalpine ecosystems

The Australian alpine and subalpine region has a range of distinctive terrestrial and freshwater ecosystem types (Figure 1). The distribution of these ecosystem types and their characteristic plant communities are driven by the distinct climate (water, snow, wind exposure), typography, soils, hydrology, and disturbance regimes (Venn et al. 2017). For example, *Snowpatch Herbfield* forms in pockets where snow persists longer than the surrounding landscape, while *Fjaeldmark/feldmark* resides only on highly exposed ridgelines and summits with periglacial conditions and strong winds. Many of these ecosystem types are naturally small and fragmented and often exist in mosaics of similar ecosystem types. For example, the water-dependent ecosystem types of *Alpine Sphagnum Bog and Associated Fen*, *Alpine-subalpine Fen*, *Damp Valley Grassland and Rushland*, and *Tasmanian Alpine Sedgeland* exist as a continuum and are distinguished by the dominant plant species and hydrology (e.g., presence and permanency of standing water). These ecosystem types can transition into one another, depending on the specific environmental and typographical conditions, such as *Alpine-subalpine Fen* degrading into *Alpine-subalpine Damp Valley Grassland and Rushland,* due to a decline in standing water causing desiccation. Another example is *Alpine-subalpine Open Grassy Heathland and Herbfield* can transition into *Alpine-subalpine Closed Heath,* if shrub cover markedly increases due to changes in disturbance regimes.

Alpine and subalpine ecosystems are home to many floral and faunal species, including endemic species. For example, the winter deciduous *Nothofagus gunnii* is endemic to *Subalpine Palaeoendemic Forest and Woodland,* and a suite of paleo-endemic coniferous shrubs – primarily *Microcachrys tetragona, Pherosphaera hookeriana, Diselma archeri* and *Podocarpus lawrencei* (Kirkpatrick 1997)*,* are endemic to *Coniferous Heath.* Other ecosystems can display substantial variation in their species composition, such as *Snowpatch Herbfield*, which is typically dominated by different varieties of short graminoids and herbs (Parry & Balmer 2017). Typical mammal species include species such as the broad-toothed rat (*Masyacomys fuscus*) and dusky antechinus (*Antechinus swainsonii*; Green & Osborne 1994). Reptiles include species such as the white-lipped snake (*Drysdalia coronoide*) and Guthega skink (*Liopholis Guthega*) (Green and Osborne, 1994), and amphibians include species such as the Critically Endangered Southern frog (*Pseudophryne corroboree)* and Northern Corroboree frog (*Pseudophryne pengilleyi*), and the Endangered Baw Baw Frog (*Philoria frosti*). Characteristic fish to mainland Australian alpine areas are typically high elevation *Galaxias* species, including *Galaxias supremus sp. nov.* (Raadik 2014), *G. findlayi* (Good

1992), and *G. olidus* (Green 2008). There are also several endemic invertebrates, such as the Cushion moth, *Nemotyla oribates*, which is dependent on cushion plants for persistence.

Key threatening processes

Australian alpine and subalpine ecosystems are affected by a wide range of interacting threatening processes. A major threatening process for Australian alpine and subalpine ecosystems is climate change. High elevations are currently experiencing the fastest rate of warming in Australia (Hennessy et al. 2003). This makes alpine ecosystems highly vulnerable to current climatic changes, given the trajectory of climate change predictions and observations (IPCC 2022). By 2100, average temperatures are expected to increase by 4–5°C and annual precipitation to decrease by 0-20% across the Australian Alps (Harris et al. 2016). Australian alpine regions have a relatively low elevation, with the highest point at 2228 m (Mount Kosciuszko, New South Wales). Therefore, there is little scope for these ecosystems to shift to higher elevations to more suitable climates.

Alpine and subalpine ecosystems are already experiencing the impacts of climate change. These include warmer average temperatures (Bhend et al. 2012), declines in precipitation, and lower depths and shorter persistence of snow cover (Green & Pickering 2009a; Bhend et al. 2012; Harris et al. 2016) There has also been an increase in the frequency of fires in many ecosystems, aligning with an increase in extreme fire weather (Richardson et al. 2021). Such environmental changes have caused shifts in biotic composition and cover. For example, shrub cover has increased in *Alpine-subalpine Open Grassy Heathland and Herbfield* (Wahren et al. 2013) and graminoids are encroaching into *Snowpatch Herbfield* (Pickering et al. 2014). More frequent, severe fires have damaged vegetation and peat in *Alpine Sphagnum Bog and Associated Fen* (Hope et al. 2005). However, other ecosystems, such as *Alpine-subalpine Closed Heath*, are relatively tolerant of burning and are predicted to expand their distribution (Williams et al. 2008) if fire frequency does not exceed the characteristic species capacity to regenerate (Enright et al. 2015).

The region is also experiencing risk caused by a range of other threats. There is a long history of livestock grazing, which has now ceased, that has caused widespread damage, especially after fires (Kirkpatrick et al. 2016). The persistent presence of feral ungulates, such as horses, deer, and pigs, has heavily degraded sensitive ecosystems, including *Alpine Sphagnum Bog and Associated Fen*, *Alpine-subalpine Streams*, and *Alpine-subalpine Open Grassy Heathland and Herbfield* (Tolsma & Shannon 2018; Driscoll et al. 2019). Energy infrastructure, such as the Snowy-Hydroelectricity Scheme (Snowy Hydro Limited 2020), has caused the physical destruction of alpine vegetation, which in turn, has altered hydrological cycles (Lawrence 2001). The effects of these threats are likely to interact with climate change (Pickering 2007). For example, *Alpine Sphagnum Bog and Associated Fen* that has previously been trampled is likely less able to tolerate and recover from a wildfire (Tolsma 2020). Climate change and climate-change driven wildfires may also increase weed invasions (Petitpierre et al. 2016). These distinct and interacting threats are contributing to risk of collapse in Australian alpine and subalpine ecosystem types.

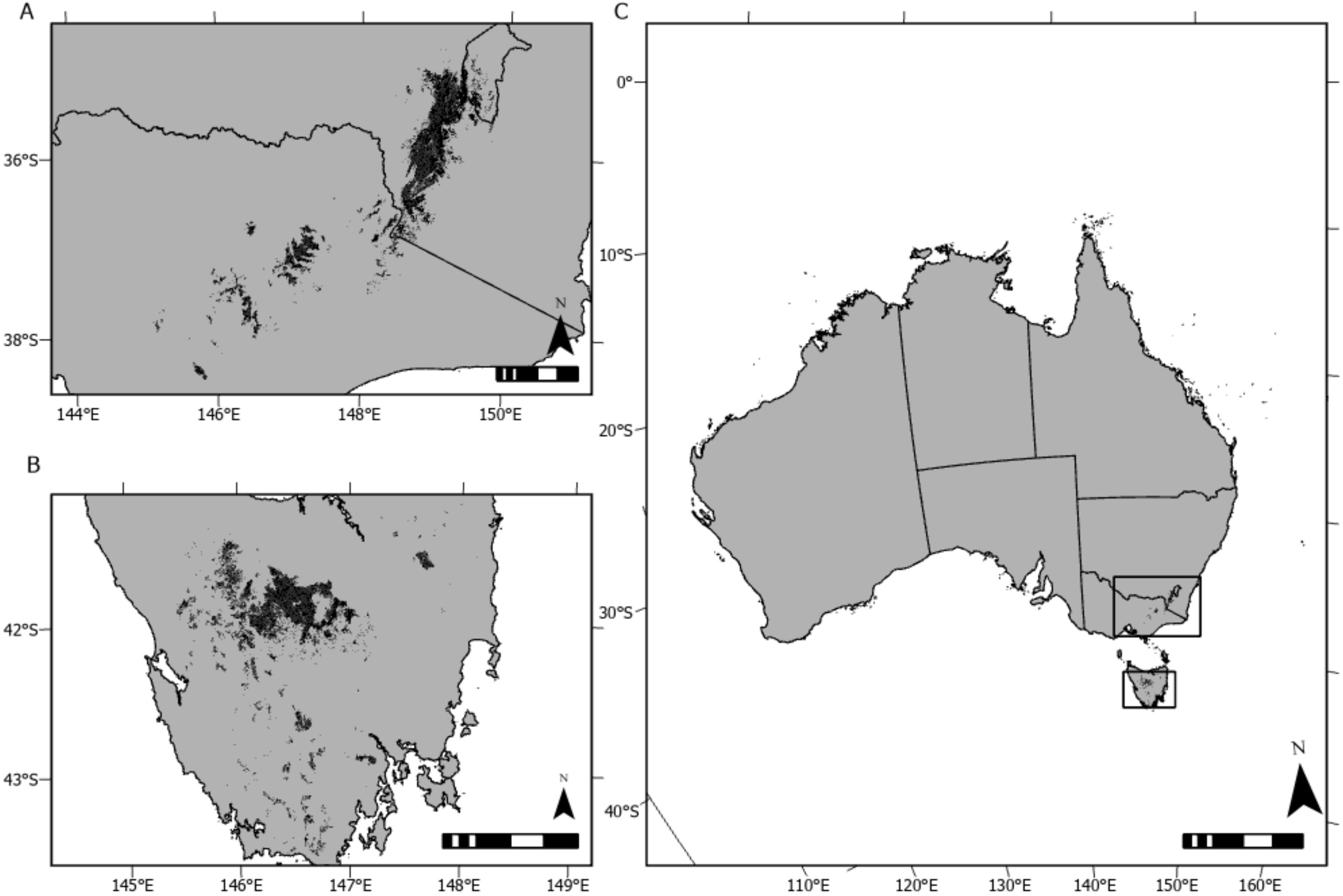


Figure 1. Map of alpine and subalpine ecosystems across Australia. Black polygons represent ecosystems. A) Mainland Australia B) Tasmania C) Australian continent.

Aims

This report documents the IUCN Red List of Ecosystems assessment for Australian alpine and subalpine ecosystem types. This project brings together leading risk assessment experts and alpine ecosystem ecologists from across Australia to:

- Develop a national typology of alpine and subalpine ecosystem types;
- Create a national map of alpine and subalpine ecosystem types; and
- Undertake IUCN Red List of Ecosystems assessments for 15 ecosystem types, including developing conceptual models, ecosystem descriptions, identifying key threatening processes, compiling existing datasets, and assessing changes in the distributions, environmental conditions, and biota for each ecosystem type.

Outputs from these assessments are intended to be used to support conservation decision making in Australian Alpine ecosystems. These assessments seek to provide information essential to achieving Australia's commitment under the Global Biodiversity Framework, including Goal A

"The integrity, connectivity and resilience of all ecosystems are maintained, enhanced, or restored, substantially increasing the area of natural ecosystems by 2050";
and related targets:
Target 1**:** "*Ensure that all areas are under participatory, integrated and biodiversity inclusive spatial planning and/or effective management processes addressing land- and sea-use change, to bring the loss of areas of high biodiversity importance, including ecosystems of high ecological integrity, close to zero by 2030, while respecting the rights of indigenous peoples and local communities."* And;

Target 2: "*Ensure that by 2030 at least 30 per cent of areas of degraded terrestrial, inland water, and marine and coastal ecosystems are under effective restoration, in order to enhance biodiversity and ecosystem functions and services, ecological integrity and connectivity."*

The outcomes from the aims of this report will provide a basis for understanding the current state and distribution of the ecosystems, which will in turn help identify changes or losses in distribution, and changes or degradation to ecological integrity and connectivity. Furthermore, this report will identify the key components of ecological integrity to each ecosystem-type, identifying key threats and the state of progression towards collapse. These will in turn help to identify recommendations to support the conservation of Australian alpine and subalpine ecosystems. Understanding the threat status and drivers of risk for these ecosystem types is required to strengthen our capacity to make evidence-based decisions for their conservation and priority setting, such as designing monitoring programs, implementing on-ground threat management, and developing policies to regulate land use

.

Introduction to the IUCN Red List of Ecosystems

The IUCN Red List of Ecosystems is the global standard for ecosystem risk assessment (Keith et al. 2013). We applied version 1.1 of the guidelines (Bland et al. 2017).

The framework uses five quantitative criteria that address the spatial and functional symptoms of collapse (Figure 2). The five criteria are as follows: change in distribution (Criterion A), current distribution and ongoing threats (Criterion B), change in the abiotic environment (Criterion C), change in the biotic features, processes, and interactions (Criterion D), and a quantitative assessment of the probability of collapse using an ecosystem simulation model (Criterion E).

Change in distribution, and the abiotic and biotic components are assessed over three standard timeframes, representing the three sub-criteria. These include

- 1) the last 50 years;
- 2a) the next 50 years;
- 2b) any 50-year period including the past, present, and future; and
- 3) since the onset of industrialised change (approximately 1750).

Each ecosystem type should be assessed using as many criteria and sub-criteria as possible.

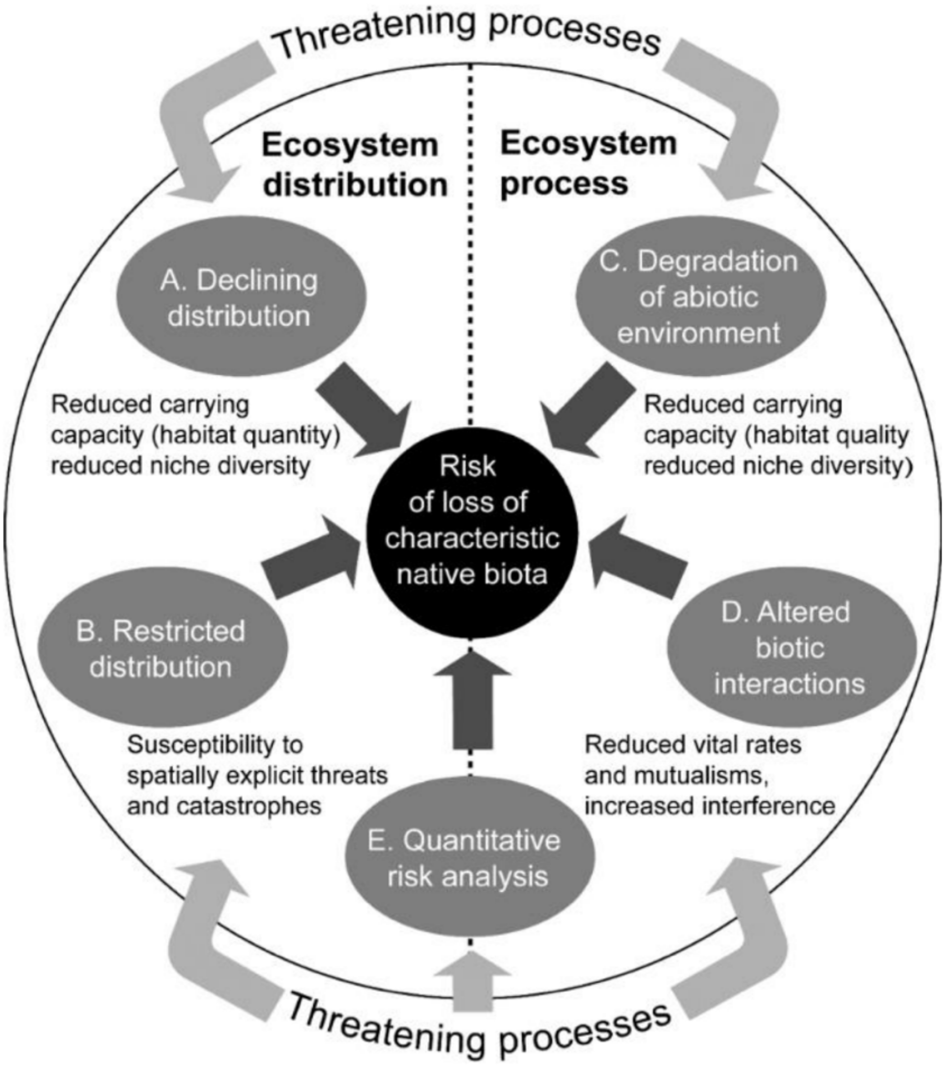


Figure 2. The IUCN Red List of Ecosystems criteria (A-E). Source: (Keith et al. 2013).

Decision thresholds are used in assessing each criterion to assign an ordinal category of risk (Figure 3; Table 2). There are six categories that represent risk status (Collapsed, CO; Critically Endangered, CR; Endangered, EN; Vulnerable, VU; Near Threatened, NT; and Least Concern, LC), one category that represents insufficient data to reliably assign a risk category (Data Deficient), and one category that represents when a criteria is not assessed (Not Evaluated).

For further details of the categories, criteria and framework, refer to the *Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria* (IUCN 2024).

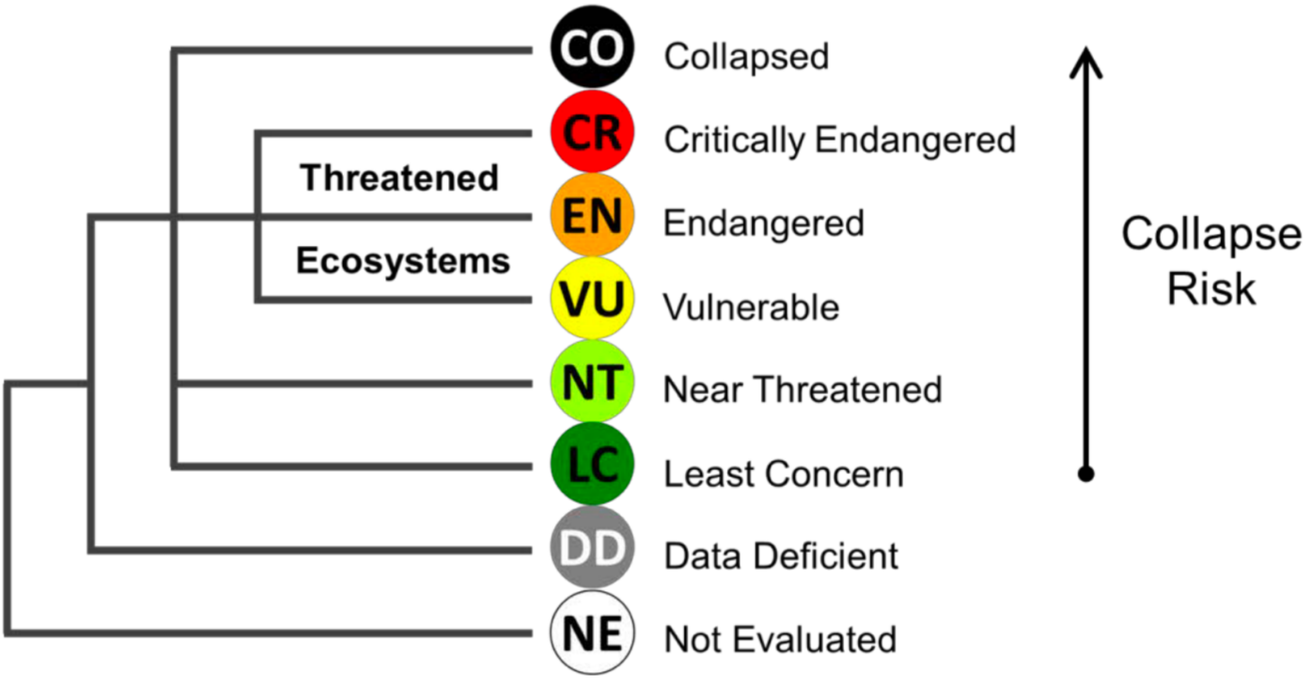


Figure 3. The IUCN Red List of Ecosystems threat categories from lowest (bottom) to highest (top) risk of collapsing. Source: (IUCN 2024).

Table 2. The IUCN Red List of Ecosystems Criteria, Version 1.1. Source: (IUCN 2024).

A. Reduction in geographic distribution over ANY of the following time periods:			
	CR	EN	VU
A1 Past (over the past 50 years)	≥ 80%	≥ 50%	≥ 30%
A2a Future (over the next 50 years)	≥ 80%	≥ 50%	≥ 30%
A2b Any 50-year period (including the past, present and future)	≥ 80%	≥ 50%	≥ 30%
A3 Historical (since approximately 1750)	≥ 90%	≥ 70%	≥ 50%

B. Restricted geographic distribution indicated by ANY OF B1, B2 or B3:			
	CR	EN	VU
B1 Extent of a minimum convex polygon (km2) enclosing all occurrences (extent of occurrence, EOO) is no larger than: AND at least one of the following (a-c): (a) An observed or inferred continuing decline in **ANY** of: i. a measure of spatial extent appropriate to the ecosystem; **OR** ii. a measure of environmental quality appropriate to characteristic biota of the ecosystem; **OR** iii. a measure of disruption to biotic interactions appropriate to the characteristic biota of the ecosystem (b) Observed or inferred threatening processes that are likely to cause continuing declines in geographic distribution, environmental quality or biotic interactions within the next 20 years	≤ 2,000 km2	≤ 20,000 km2	≤ 50,000 km2
(c) Ecosystem exists at:	1 threat-defined location	≤ 5 threat-defined locations	≤ 10 threat-defined locations
B2 The number of 10 × 10 km grid cells occupied (area of occupancy, AOO) is no more than: **AND** at least one of a-c above (same as for B1).	≤ 2	≤ 20	≤ 50
B3 The number of threat-defined locations is very small (generally fewer than 5) **AND** prone to the effects of human activities or stochastic events within a very short time period in an uncertain future, and thus capable of Collapse or becoming Critically Endangered (CR) within a very short time period (B3 can only lead to a listing as **VU**).			VU

C. Environmental degradation over ANY of the following time periods:	Relative severity (%)			
C1 The past 50 years, based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table	Extent (%)	≥ 80	≥ 50	≥ 30
	≥ 80	CR	EN	VU
	≥ 50	EN	VU	
	≥ 30	VU		
C2 **C2a.** The next 50 years, based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table; **OR** **C2b**. Any 50-year period including the past, present and future, based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	Extent (%)	≥ 80	≥ 50	≥ 30
	≥ 80	CR	EN	VU
	≥ 50	EN	VU	
	≥ 30	VU		
C3 Since 1750 based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	Extent (%)	≥ 90	≥ 70	≥ 50
	≥ 80	CR	EN	VU
	≥ 50	EN	VU	
	≥ 30	VU		

D. Disruption of biotic processes or interactions over ANY of the following time periods:	Relative severity (%)			
D1 The past 50 years, based on change in a <u>biotic</u> variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table	Extent (%)	≥ 80	≥ 50	≥ 30
	≥ 80	CR	EN	VU
	≥ 50	EN	VU	
	≥ 30	VU		
D2 **D2a.** The next 50 years, based on change in an <u>biotic</u> variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table; **OR** **D2b**. Any 50-year period including the past, present and future, based on change in a <u>biotic</u> variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	Extent (%)	≥ 80	≥ 50	≥ 30
	≥ 80	CR	EN	VU
	≥ 50	EN	VU	
	≥ 30	VU		
D3 Since 1750 based on change in a <u>biotic</u> variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	Extent (%)	≥ 90	≥ 70	≥ 50
	≥ 80	CR	EN	VU
	≥ 50	EN	VU	
	≥ 30	VU		

E. Quantitative analysis that estimates the probability of ecosystem collapses to be:	
CR	≥ 50% within 50 years
EN	≥ 20% within 50 years
VU	≥ 10% within 100 years

Mountains in shadow, Kosciuszko National Park. Photo: Australian Alps collection – Parks Victoria.

Methods

Overview

This is the first national assessment of Australian alpine and subalpine ecosystem types. To complete the project, we held a three-day workshop in Melbourne, Australia in December 2019 with 26 alpine ecosystem ecologists and experts in the IUCN Red List of Ecosystems from academic institutions, state and federal governments, and non-for-profit groups across New South Wales, Victoria, the Australian Capital Territory, and Tasmania. During the workshops, the team refined the typology of ecosystem types and divided into small groups to draft written descriptions and conceptual models for each ecosystem type, identify and collate suitable datasets to assess the criteria, and identify relevant indicators to assess. An additional two-hour virtual workshop was conducted in October 2020 to select indicators to assess the potential risks posed by climate change. The assessments were completed after the workshops via email and in virtual meetings.

Ecosystem typology

We developed a list of 17 candidate ecosystem types to assess based on Venn et al. (2017). During the workshop series, the experts reviewed the candidate ecosystem types. The final list of 15 ecosystem types was decided by consensus among the experts, and resulted in slight departures from Venn et al. (2017); some ecosystem types were grouped together in our study to form broader classifications where they had key similar processes driving ecosystem function, structure and composition (e.g., damp valley grasslands and damp herbfields were grouped into *Subalpine Damp Valley Grassland and Rushland*) or similar types occurred on the mainland and Tasmania (*Alpine-subalpine Open Grassy Heathland and Herbfield*). *Alpine-subalpine Sphagnum Bog and Associated Fen* was assessed as a national unit matching a Threatened Ecological Community currently listed under Australian legislation (Threatened Species Scientific Committee 2009), as well as assessed separately for the mainland and Tasmania's extents due to compositional differences. We cross-references the 15 ecosystem types to six functional groups across three realms (terrestrial, freshwater-terrestrial transitional, and freshwater) in the IUCN Global Ecosystem Typology (Keith et al. 2022a), using expert judgement based on key characteristics and properties of the ecosystem types and ecosystem function groups.

Ecosystem mapping

To develop robust and accurate maps of national alpine and subalpine ecosystem distributions, we used available data for each ecosystem and applied elevation thresholds to refine ecosystem boundaries. Spatial analyses were performed using the R programming environment version 4.1.2 (R Core Team 2023), utilising the spatial analysis packages *terra* (Hijmans 2025) and *sf* (Pebesma & Bivand 2023).

Data sources and preparation

We developed a synthesis ecosystem map for the alpine-subalpine bioregion, representing the distributions of each of the assessed ecosystems, by integrating various data sources to create representative ecosystems. Ecosystems-specific data sources are shown in Table 3. To develop a synthesis ecosystem map, we combined the source ecosystems based on the rules in Table 3. We cut the source ecosystem layers using the minimum threshold elevations in Table 3 (based on expert judgement for each ecosystem type), using the Digital Elevation Model (DEM) (Gallant et al. 2019). We used state vegetation maps as the base layer (see list below). However, because some ecosystems had been mapped independently at finer scales or using classifications that corresponded more accurately to the ecosystem descriptions, these were given precedence over base vegetation layers, where mapped distributions overlapped. The order was agreed by experts during the mapping process (Table 3).

The data used as a base layer were primarily from state vegetation maps, in addition to national datasets for freshwater ecosystems, bioregions and elevation (see Table 3 for all datasets used):

- Australia wide Digital elevation layer Ga.gov.au 1 second DEM (30m res) (Crossman 2025)(https://pid.geoscience.gov.au/service/ga/89718)
- Australian bioregions layer' (Gallant et al. 2019) (https://pid.geoscience.gov.au/dataset/ga/72759)
- For New South Wales (NSW) and the Australian Capital Territory (ACT):
    - Forest Ecosystems VISID 3858 (DCCEEW 2011a) for 'current' (2005) https://datasets.seed.nsw.gov.au/dataset/c13950e1-9afd-4aa6-8064-8783f3d4fd57
    - Forest Ecosystems VISID 3859 (DCCEEW 2011b) for 'pre-1750 historical distributions' (https://datasets.seed.nsw.gov.au/dataset/forest-ecosystems-native-vegetation-of-the-southern-forests-south-east-highlands-australian-38a92)
    - Kosciuszko National Park Resorts Vegetation Thredbo 2003 Vis_ID 4841 (DCCEEW 2003a) https://datasets.seed.nsw.gov.au/dataset/kosciuszko-national-park-resorts-vegetation-thredbo-2003-vis_id-4841
    - Kosciuszko National Park Resorts Vegetation Perisher 2003 VIS_ID 4840 (DCCEEW 2003b) https://datasets.seed.nsw.gov.au/dataset/kosciuszko-national-park-resorts-vegetation-perisher-2003-vis_id-4840
    - Kosciuszko National Park Resorts Vegetation Mount Selwyn 2003 VIS_ID 4839 (DCCEEW 2003c) https://datasets.seed.nsw.gov.au/dataset/kosciuszko-national-park-resorts-vegetation-mount-selwyn-2003-vis_id-4839
    - Kosciuszko National Park Resorts Vegetation Charlotte Pass 2003 VIS_ID 4838 (DCCEEW 2003d) https://datasets.seed.nsw.gov.au/dataset/kosciuszko-national-park-resorts-vegetation-charlotte-pass-2003-vis_id-4838
    - Kosciuszko National Park Resorts Vegetation Bullocks Flat 2003 VIS_ID 4837 (DCCEEW 2003e) https://datasets.seed.nsw.gov.au/dataset/kosciuszko-national-park-resorts-vegetation-bullocks-flat-2003-vis_id-4837
- ACT: ACT Vegetation Map 2018 (ACT Government 2018) (https://services1.arcgis.com/E5n4f1VY84i0xSjy/arcgis/rest/services/ACTGOV_Vegetation_Map_2018/FeatureServer).
- For Victoria: Native Vegetation - Modelled 2005 Ecological Vegetation Classes (EVC) distribution (DELWP 2018a) (https://datashare.maps.vic.gov.au/search?q=uuid%3D58e3ca32-e951-5bfa-9974-728cea56a14e).
- For Victoria: Native Vegetation - Modelled 1750 Ecological Vegetation Classes (EVC) distribution (DELWP 2018b); (https://datashare.maps.vic.gov.au/search?q=uuid%3D853bda8a-a680-5ec4-9cbe-8a33b51eee2b)
- For Tasmania: TASVEG 4.0 digital Tasmanian vegetation map (DPIPWE 2020)(current only); (https://nre.tas.gov.au/conservation/development-planning-conservation-assessment/planning-tools/monitoring-and-mapping-tasmanias-vegetation-(tasveg)/tasveg-the-digital-vegetation-map-of-tasmania).

All data were reprojected to the MGAzone55 coordinate system to ensure consistency across datasets. Because of mapping errors due to changes in projection, different data sources and resolutions, assumptions above boundaries, and other sources, polygon fragments were created during the development of the synthesis map. By checking a subset of these against high resolution satellite imagery, we found that they related to mapping errors (such as boundary misalignment) rather than genuine small occurrences. These caused problems is displaying and analysing the maps, especially for Criterion B. Therefore, we removed polygons that were smaller than 5m^2 in all ecosystems, except for those where small patches naturally occurred (such as *Snowpatch Herbfield and Alpine Sphagnum Bog and Associated Fen*).

Table 3. Datasets use to produce national map of alpine-alpine ecosystem types. Order of overlapping ecosystem distributions to create mutually exclusive synthesis map (where lowest form the base map).

Order	Ecosystem type	NSW	ACT	Victoria	Tasmania
1 (highest)	Snowpatch herbfield	≥1500m (Costin et al. 2000) aerial photography in 2015 and 2017 (McDougall, unpublished data) ski resort vegetation mapping.	≥1500m (Costin et al. 2000)	≥1500m EVC dataset (DELWP 2018a): 1012 and 1014 (DELWP 2018a). Minor edits to Mount Nelse (EVC 1014)	≥1100m (Kirkpatrick & Marsden-Smedley 2014)
2	Alpine-subalpine Lakes	≥1500m national surface hydrology, using lakes >1ha and not anthropogenic in origin (Crossman & Li 2015)	≥1500m See text for detail (Crossman & Li 2015)	≥1500m See text for detail (Crossman & Li 2015)	≥915m See text for detail (Crossman & Li 2015)
3	Fjaeldmark/feldmark	≥2000m Kosciuszko national park resorts vegetation mapping (VISID 4841, 4840, 4839, 4838, 4837, 4842). (Costin et al. 1979, 2000) (VISID 4842): Feldmark Spp 1, Feldmark Spp 2, Heaths Short, Tall Alpine Herbfield Spp 1, Tall Alpine Herbfield Spp 2, and Short Alpine Herbfield.	≥2000m Same as NSW	NA	≥900m (Kirkpatrick & Marsden-Smedley 2014)
4	Alpine Sphagnum Bog and Associated Fen	≥1000m VISID3858: "Montane/Sub-alpine Sedge Swamps" (Hope et al. 2012): "Alpine Sphagnum Shrub Bog", "Empodisma minus restiad moor/Sub-Alpine Sphagnum Shrub Bog", "Medium Altitude Shrub Bog", "Sub-Alpine Sphagnum Shrub Bog"	≥1000m ACTmap (ACT Government 2018): UMC_ID u193	≥1000m EVC dataset (DELWP 2018a): EVC 171 and parts of EVC 44 assigned by experts using high resolution aerial photography; supplemented by Tolsma/Sutter aerial photo (2004-2018) interpretation compiled in 2019 (unpublished), drawing on previous maps published of the Baw Baw Plateau (Roberts 1996), Walsh, Gullen and White (Walsh et al. 1984), and the Bogong High plans (McDougall 1982). Attribute data records these as: "Sub-alpine Wet Heathland", "Alpine Hummock Peatland", "Alpine Fen", "Alpine fen", "Alpine Heath Peatland"	≥800m TASVEG3.0: MSP (Tasmanian Goverment 2014) TASVEG 4.0: ASP (DPIPWE 2020)
5	Alpine-subalpine Fen	≥1000m (Hope et al. 2012): vegetation group “Carex Fen”	≥1000m ACTmap (ACT Government 2018):	≥1000m	≥800m TASVEG 4.0 (DPIPWE 2020):

			UMC_ID: ("a8", "a9", "e59")		VEGCODE: AHL, ASF
6	Coniferous Heath	See main text. NSW boulderfield mapping (Broome et al. 2013)	ACTmap (ACT Government 2018): UMC_ID “a54”	EVC dataset (DELWP 2018a): EVC 156 Mountain pygmy possum habitat (Heinze & Harvey 2006; Harvey 2007; ARMB 2011)	TASVEG 4.0 (DPIPWE 2020): Vegcode HCH
7	Damp Valley Grassland and Rushland	≥1200m VISID3858: Sub-alpine Wet Herb-Grassland-Bog” “Empodisma minus restiad moor” (Hope *et al* 2012); https://datasets.seed.nsw.gov.au/dataset/peat-forming-bogs-and-fens-of-the-snowy-mountains)	≥1200m ACTmap (ACT Government 2018): UMC_ID “a2”	≥1200m EVC dataset (DELWP 2018a): EVC 1002 and parts of EVC 44	≥1100m TASVEG 4.0 (DPIPWE 2020): 4.0HSE Digital aerial photography interpreted by Jamie Kirkpatrick in 2019, represented by vegcodes GPH and MGH
8	Alpine-subalpine Closed Heath	≥1100m VISID 3858: "Alpine Rocky Low Open Heathland" and "Montane - Sub-Alpine Dry Rocky Shrubland.	≥1100m ACTmap (ACT Government 2018): “a33” and “g36”	≥1100m EVC dataset (DELWP 2018a): 1003, 211, 208, 210, 42, 1105, 1014, 1000, parts of EVC 44	NA
9	Alpine-subalpine Open Grassy Heathland and Herbfield	≥1100m VISID 3858: "Sub-alpine Herbfield", "Sub-alpine Dry Herb-Grassland", "Alpine Wet Herbfield", "Short Alpine Herbfield".	≥1100m ACTmap (ACT Government 2018): “a14”	≥1100m EVC dataset (DELWP 2018a): 1004, 1001, 317, 1005, 206 and parts of EVC44.	≥900m TASVEG 4.0 (DPIPWE 2020): GPH
10	Alpine-subalpine Woodland and Forest	≥1350m VISID 3858 (with no elevation cut): "Scabby Range Dry Shrub Woodland", "Montane Dry Shrub-Herb-Grass Forest", "Sub-alpine Dry Shrub-Herb-Grass Woodland", "Sub-alpine Shrub-Grass Woodland", "Sub-alpine Dry	≥1350m ACTmap (ACT Government 2018): "u158", "u207"	≥1000m EVC dataset (DELWP 2018a): 43 and parts of EVC 44	≥800m TASVEG 4.0 (DPIPWE 2020): "DCO" "DGW"

		Shrub-Herb Woodland", "Montane Acacia-Dry Shrub-Herb-Grass Forest" VISID 3858 (cut to 1350m elevation and above): "Western Sub-alpine Moist Shrub Forest", and "Western Montane Moist Shrub Forest"			
11	Subalpine Palaeoendemic Forest and Woodland				TASVEG 4.0 (DPIPWE 2020): "RPF", "RFS", "RKS", "RPP", "RKP", "RPW", "RKF" (with no elevation cuts)
11	Cushion Moorland				≥800m TASVEG 4.0 (DPIPWE 2020): “HCM”
11	Tasmanian Alpine heath				≥800m TASVEG 4.0 (DPIPWE 2020): “HHE", "HHW", "SHS", "SSW"
11	Tasmanian Alpine Sedgeland				TASVEG 4.0 (DPIPWE 2020): “HSE”, “MGH”, and “HSW”
12	Alpine-subalpine Streams	≥1500m Water courses and watercourse lines were extracted from national surface hydrology (Crossman & Li 2015).	See NSW text	≥1350m Water courses and watercourse lines were extracted from national surface hydrology (Crossman & Li 2015).	≥915m Water courses and watercourse lines were extracted from national surface hydrology (Crossman & Li 2015).

Ecosystem Distribution Analysis (Criteria A and B)

The *redlistr* package (Lee et al. 2019) was used to calculate three key metrics: the Area of Distribution (Criterion A), Area of Extent of Occurrence (Criterion B) and Area of Occupancy (Criterion B). To facilitate this analysis, ecosystem layers were first converted to a raster using the *fasterize* function, at a resolution of 50m.

Area of Distribution

The *getArea* function was applied to calculate the total area covered by each ecosystem. This function provides a measure of the spatial extent of ecosystems.

Area of Extent of Occurrence

To measure the geographical spread of each ecosystem, we employed the *getAreaEOO* function. This function calculates the smallest area that can contain all occurrences of an ecosystem, offering insights into its overall geographical presence and the extent of occurrence.

Area of Occupancy (Grid uncertainty analysis)

A critical aspect of spatial analysis in conservation assessment is accounting for uncertainty. The *getAOO* function was used to assess the Area of Occupancy (AOO) with specific respect to grid uncertainty. For this analysis, we defined a grid size of 10,000m and explored various minimum occupancy scenarios: 1%, 0.1%, 0.01%, and no minimum. These scenarios allowed us to understand how varying scenarios of occupancy affect the calculated area of occupancy.

To facilitate this analysis, our ecosystem layers were rasterized using the *fasterize* function at a resolution of 50m. This step ensured that our spatial data were appropriately formatted for grid-based analysis, allowing for accurate calculation of the AOO under different minimum occupancy rules.

Ecosystem description and Conceptual models

During the three-day workshop, the experts were divided into five small working groups and each group was allocated two or three ecosystem types in their field of expertise. Each group also had at least one member with expertise in the IUCN Red List of Ecosystems. Each group developed a conceptual model and description for their allocated ecosystem types based on information compiled from literature searches and the experts' knowledge. The conceptual models were then simplified to only capture the key features and processes and the terminology standardised for consistency across ecosystem types. Each ecosystem description included information on the typological classification, distinction from similar ecosystem types, distribution, abiotic environment, characteristic native biota, interactions and processes, key threatening processes, and ecosystem collapse.

IUCN Red List of Ecosystems assessment

For each ecosystem type, we assessed all the IUCN Red List of Ecosystems criteria. Here, we outline the general methods used in the process. Specific methods for each ecosystem type are detailed in assessments below.

Criterion A

We assessed changes in the extent of each ecosystem type over time using publicly available spatial data and published estimates of area or change in area.

Due to data limitations, the Criterion A3 (historical loss pre-1750) analysis was only conducted on three ecosystems: *Alpine-subalpine Woodland and Forest, Alpine-subalpine Open Grassy Heathland and Herbfield, Alpine-subalpine Closed Heath,* and *Alpine-subalpine Damp Valley Grasslands and Rushlands,* as their vegetation classes were primarily sourced from SEED (NSW) and EVCs (VIC) (with distributions from ACT data or Hope et al. (2012) kept constant) (see Table 3 for datasets), which include modelled historical distributions. We were unable to conduct complimentary analysis for other ecosystems as they were compiled using datasets which did not have equivalent historical datasets. Because the full potential distribution from underlying datasets were used, rather than the synthesis map (which did not allow overlap between ecosystem types from differing data sources), the estimated current area in A3 analyses may differ from the actual distribution reported.

Because of uncertainties around distribution, we ran two scenarios to assess changes in ecosystem distributions:

1. **Scenario 1 (Conservative Approach):** Utilized identical elevation cuts for both historical and extant distributions. This scenario assumes no elevation change in ecosystem boundaries over time, providing a baseline for minimal distribution change.

2. **Scenario 2 (Non-conservative Approach):** Adjusted the elevation cuts for historical distributions to be 50m lower than their extant counterparts. This adjustment accounts for an estimated upslope movement of communities at a rate of 1m per year since 1950, reflecting changes due to ambient temperature increases. The elevation rate change is based on findings by Slatyer (2010) and Hennessy et al. (2003, 2007), with the year 1950 chosen based on accelerated climate change patterns reported by BOM & CSIRO (2020).

To quantify the changes in ecosystem distributions, we then calculated net and proportional loss. Net loss was calculated as the total area of gain subtracted from the total area of loss. A proportional net loss was then calculated as the net loss divided by the historical distribution area, providing a measure of loss relative to the original distribution size.

Criterion B

Using the current distribution maps, we calculated the Extent of Occurrence (EOO) and Area of Occupancy (AOO) using the R package *redlistr* (Lee et al. 2019). We did not apply the 1% rule when calculating the AOO because many of the ecosystem types have a naturally small, fragmented distribution and thus small patches can contribute meaningfully to risk-spreading (Bland et al. 2017). To assess the presence of ongoing declines or threats and the number of threat-defined locations, we reviewed available literature and consulted with experts.

Criterion C

We estimated change in the characteristic environmental conditions of each ecosystem type using a range of ecosystem-specific indicators. We reviewed the literature and relied on expert knowledge of the ecosystem types to identify suitable ecosystem-specific indicators and identify and collate data sources to assess change. Sufficient data were available to assess Criterion C for eight ecosystem types. The indicators assessed include fire occurrence or frequency, bare ground cover, snowmelt date, and moisture balance. We used common methods in the analysis of fire occurrence or frequency across ecosystem types. These are detailed below.

Fire analysis

The occurrence of fire was assessed for *Coniferous Heath*, *Subalpine Palaeoendemic Forest and Woodland*, and the Tasmanian extent of *Alpine-subalpine Open Grassy Heathland and Herbfield*. Change in the frequency of fires was assessed for *Alpine-subalpine Closed Heath*, *Alpine-subalpine Woodland and Forest, Tasmanian Alpine Heath,* and the mainland extent of *Alpine-subalpine Open Grassy Heathland and Herbfield*. Future fire was assessed for the mainland extents of *Coniferous Heath Alpine-subalpine Closed Heath*, *Alpine-subalpine Woodland and Forest* and *Alpine-subalpine Open Grassy Heathland and Herbfield*. Future fire analyses were also performed for *Alpine Sphagnum Bog and Associated Fen* and *Alpine-subalpine Fen*, and paired to future of occurrence of drought analyses (detailed in the assessments of each ecosystem type).

Recent and historical change (sub-criteria C1 and C3)

To assess past change in these fire indicators (sub-criteria C1 and C3), we collapsed historical records (as shapefiles) of bushfires and planned burns for New South Wales and the Australian Capital Territory (NSW Goverment 2022), Tasmania (Tasmanian Government 2022), and Victoria (Victorian Government 2022a). The dataset for New South Wales includes the Australian Capital Territory, and covers the 1902/1903 to 2020/2021 fire seasons (NSW Goverment 2022). The Victorian dataset includes fires occurring between the 1902/1903 to 2019/2020 fire seasons, primarily for fires on public lands (Victorian Government 2022a). The Tasmanian fire history dataset includes fires in the 1960/1961 and 2019/2020 fire seasons in regions greater than 600 m above sea level (Tasmanian Government 2022). The datasets for New South Wales, the Australian Capital Territory and Victoria were merged to create a single file for the mainland. Where fire records from the different data sources overlapped in each fire season, the records were merged. Fire seasons are defined as fires that occur between July in one year and June in the next year.

To define the area burnt across the ecosystem types over the past 50 years (C1) or historically (C3), we overlayed the fire history layers and extracted areas where each ecosystem type had been burnt. For each sub-criteria, we calculated the area of the ecosystem type that had burnt within the timeframe. For the ecosystem types where fire occurrence was the indicator assessed, these data were used to define the extent of degradation, assuming the severity was 100% (collapsed). For ecosystem types where fire frequency was assessed, we generated timeseries by calculating the number of times and area burnt for intervals that represent the collapse threshold (depending on the ecosystem type). We used these values to calculate the mean annual probability of burning based on the spatially weighted mean area of the ecosystem type burnt at each frequency in each timeframe.

Future change (sub-criterion C2)

To estimate collapse risk from future change in the frequency of fires, we obtained fire simulations for mainland Australian alpine and subalpine ecosystems for 2060 to 2079 from the landscape fire modelling framework ‘Fire Regime and Operations Tool’ (FROST). FROST encapsulates three modules: a weather, ignition, and fuel module, which informs a fire event simulator ‘PHOENIX

RapidFire' (Tolhurst et al. 2008) in combination with Bayesian network models (Penman et al. 2015).

The weather module incorporates future weather predictions on an hourly scale, from the 'NARCliM' project (Evans et al. 2014). Daily weather informs the number of ignitions predicted, and hourly weather informs the predictions of fire behaviour once fires are ignited (Mccoll-Gausden et al. 2022). The NARCliM climate projections occur at a 10-km resolution, and include surface air temperature, surface specific humidity, near-surface wind speed and direction, surface wind speed, and surface pressure (Evans et al. 2014). We selected four Regional Climate Models based on accuracy of predicting the Forest Fire Danger Index (Clarke & Evans 2019; Mccoll-Gausden et al. 2022): the ECHAM5 Regional Climate Model (RCM) 1 (ECHAM-R1), ECHAM5 RCM2 (ECHAM-R2), CSIRO Mk3 RCM1 (CSIRO-R1), and CSIRO MK3 RCM3 (CSIRO-R3). These climate models represent relatively dry conditions (Mccoll-Gausden et al. 2022).

The ignition module is informed by a static measure of proximity to roads and the density of housing (Mccoll-Gausden et al. 2022). We used a 'no management' scenario that does not incorporate prescribed burning into estimates. The number and time of ignitions per day are simulated using a Bayesian network that also accounts for historical ignitions (Mccoll-Gausden et al. 2022). The fuel module predicts fuel hazard for native fuels and non-native fuels separately, in four strata: surface, near-surface, elevated, and bark. Definitions of the fuel hazard follow existing definitions of fuel hazard (Hines et al. 2010), and are predicted using random forest models derived from soil, climate and time since fire (McColl-Gausden et al. 2020).In combination, these three modules inform the number and timing of ignitions per day, the potential for fire spread and behaviour and interaction with concurrent fires, and the consumption and accumulation of fuel (Mccoll-Gausden et al. 2022). Fires across the period are simulated at a 180 m resolution.

We used FROST to create 100 replicate simulations of the 20-year period (2060-2079). For each simulation, we extracted a spatial raster that displayed the number of fires burnt within the time period, for each 180 m^2 cell. For each climate scenario, we calculated the percentage of the ecosystem type (based on the number of cells) that burnt at a frequency that exceeded the collapse threshold (i.e., where relative severity is 100%).

Criterion D

We estimated change in the characteristic native biota of each ecosystem type using a range of ecosystem-specific indicators. We reviewed the literature and relied on expert knowledge of the ecosystem types to identify suitable ecosystem-specific indicators and identify and collate data sources to assess change. Data were only available to assess indicators for criterion D for three ecosystem types, including *Alpine-subalpine Open Grassy Heathland and Herbfield*, *Alpine-subalpine Closed Heath*, and *Snowpatch Herbfield*, and constituting analysis of shrub and graminoid cover. Details of each analysis are outlines in the assessments below.

Criterion E

We conducted literature searches and consulted with experts to identify suitable models. We were only able to assess Criterion E for one ecosystem type – *Alpine-subalpine Streams*. We used existing models of change in rainfall and runoff based on 42 CMIP 5 and 37 CMIP6 global climate models (GCMs) for Australia (Zheng et al. 2024) to scale hydrologic gauge data to provide estimates future water flows. All ecosystem types were listed as Data Deficient under this criterion due to the lack of suitable models. There was a potentially suitable model was identified for *Alpine-subalpine Lakes*, but it could not feasibly be used in this assessment due to time limitations and lack of data to suitably adapt the model for alpine conditions. Details of the analysis is outlined in the assessment below.

Photo: Susanna Venn.

Australian alpine and subalpine ecosystem types

The Australian national assessment of alpine and subalpine ecosystems includes 15 ecosystem types/units (Table 4). These ecosystem types correspond to five functional groups in the IUCN Global Ecosystem Typology (Keith et al. 2022a) within the Terrestrial biome, Freshwater biome and Freshwater-Terrestrial transitional biome. We also aligned these with an existing typology of alpine and subalpine ecosystems, and regional based classifications used in government planning, including Vegetation Groups, and Ecological Vegetation Classes.

Table 4. Relationship between the ecosystem units in this project, the IUCN global ecosystem typology (Keith et al. 2022a), and typology of alpine and subalpine ecosystems of Australia (Venn et al. 2017). Shading used to identify ecosystem units with the same functional group in the IUCN global ecosystem typology.

Ecosystem Units	**IUCN Global Ecosystem Typology**	**Alpine, Sub-alpine and Sub-Antarctic Vegetation of Australia**
Tasmanian Palaeoendemic Forest and Woodland	T2.3 Oceanic temperate rainforests	Deciduous Heath
Alpine-subalpine Woodland and Forest	T4.4 Temperate woodlands	Sub-alpine Woodland
Alpine-subalpine Open Grassy Heathland and Herbfield	T6.4 Temperate alpine meadows and shrublands	Tussock grasslands Tall alpine herbfield Open heathland Limestone grassland Marsupial lawn
Alpine-subalpine Closed Heath	T6.4 Temperate alpine meadows and shrublands	Closed heathland
Coniferous Heath	T6.4 Temperate alpine meadows and shrublands	Podocarp Shrubland Coniferous heath
Fjaeldmark/feldmark	T6.4 Temperate alpine meadows and shrublands	Feldmark/Fjaeldmark Mat heath
Snowpatch Herbfield	T6.4 Temperate alpine meadows and shrublands	Snowpatch feldmark Short-turf snowpatch Short alpine herbfield
Cushion Moorland	T6.4 Temperate alpine meadows and shrublands	Bolster heath
Tasmanian Alpine heath	T6.4 Temperate alpine meadows and shrublands	Alpine sedgeland
Tasmanian Alpine Sedgeland	T6.4 Temperate alpine meadows and shrublands	Coniferous heath
Damp Valley Grassland and Rushland	T6.4 Temperate alpine meadows and shrublands	Valley grasslands Alpine sedgeland

Alpine Sphagnum Bog and Associated Fen	TF1.5 Boreal, temperate and montane peat bogs	Bog Wet Heath Short alpine herbfield Damp herbfields
Alpine-subalpine Fen	TF1.5 Boreal, temperate and montane peat bogs	Fen
Alpine-subalpine Streams	F1.3 Freeze-thaw rivers and streams	Aquatic vegetation Gravelly Stream herbfield
Alpine-subalpine Lakes	F2.4 Freeze-thaw freshwater lakes	Aquatic vegetation

Ecosystem descriptions and assessments

Mount Feathertop covered in snow, Australian Alps collection. Photo: Parks Australia.

Terrestrial biome

Photos: Susanna Venn.

Subalpine Palaeoendemic Forest and Woodland

Authors

David Keith, Jessica Rowland

Reviewers

Jayne Balmer and José. R Ferrer-Paris

Biome

T2 Temperate-boreal forests and woodlands

Functional group

T2.3 Oceanic temperate rainforests

IUCN status

Vulnerable (Vulnerable-Endangered)

Pencil pine forest below the Temple, Walls of Jerusalem, Tasmania. Photo: Peter Dombrovskisi (1990).

Assessment Summary

Subalpine Palaeoendemic Forest and Woodland is dominated by a unique suite of palaeoendemic coniferous trees or deciduous angiosperm trees, which are highly sensitive to fire and drought. They are restricted to long-unburnt fire refuges in central, western, and southern Tasmania. The status of the ecosystem type is **Vulnerable** (plausible bounds **Vulnerable – Endangered**) due to its limited extent of occurrence (sub-criterion B1) and high possibility of fire causing the ecosystem type to collapse or become Critically Endangered within a very short period (sub-criterion B3) (Table 5).

Table 5. Summary of the Red List of Ecosystems assessment of the *Subalpine Palaeoendemic Forest and Woodland*. Category ranges in parentheses reflect uncertainty in assessment outcome under the corresponding criteria.

Criteria	**A**	**B**	**C**	**D**	**E**	**Overall**
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	VU (VU-EN)	DD	DD	DD	VU (VU-EN)
Sub-criterion 2 *A, C, D: 50-year period including present & future* *B: AOO*	VU	LC	DD	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	NT (LC-EN)	VU	DD	DD		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria

Ecosystem Description

Ecosystem Classification

In the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022a), this sub-global ecosystem type (Level 6) belongs to Ecosystem Functional Group *T2.3 Oceanic temperate rainforests* within the *Temperate-boreal forests and woodlands* biome.

Within Australia, *Subalpine Palaeoendemic Forest and Woodland* includes deciduous Heath (Kirkpatrick 1983, 1997; Venn et al. 2017) and the following conifer-dominated communities: *Athrotaxis cupressoides* - *Nothofagus gunnii* short rainforest (RPF), *Athrotaxis selaginoides* - *Nothofagus gunnii* short rainforest (RKF), *Athrotaxis selaginoides* subalpine scrub (RKS), *Nothofagus gunnii* rainforest and scrub (RFS), *Athrotaxis cupressoides* rainforest (RPP), *Athrotaxis selaginoides* rainforest (RKP) and *Athrotaxis cupressoides* open woodland (RPW) (Kitchener & Harris 2013).

The palaeoendemic dominant species (Jordan et al. 2016)are of world heritage significance and thus protected under the *Environment Protection and Biodiversity Conservation Act 1999*.

Distinction from similar ecosystem types

Subalpine Palaeoendemic Forest and Woodland is structurally similar to *Subalpine Woodland and Forest*. These two ecosystem types can be distinguished by the characteristic plants. *Subalpine Palaeoendemic Forest and Woodland* is defined by the winter deciduous *Nothofagus gunnii* and gymnosperms in the genus *Athrotaxis*, which date back to the Cretaceous (Jordan et al. 2016), and several other clades such as *Diselma archeri* in the understorey. The understorey also includes alpine shrubs including species of *Orites, Richea, Olearia, Coprosma* and *Olearia*, while the ground layer includes *Restionaceae* (cord rushes) and sometime *Sphagnum cristatum*. *Subalpine Woodland and Forest* is dominated by low trees (< 12 m) such as *Eucalyptus coccifera*, *E. gunnii*, and *E. subcrenulata* in Tasmania (Kitchener & Harris 2013; Venn et al. 2017) and by *E. pauciflora* subsp. *niphophila* and *Eucalyptus pauciflora* subsp. *pauciflora* on the mainland (Venn et al. 2017). In both regions, the understorey comprises grasses, herbs, and shrubs.

Distribution

Subalpine Palaeoendemic Forest and Woodland is located between 145.46° and 147.15° longitude and between -41.45° and -43.54° latitude (Figure 4). The ecosystem type is confined to three Tasmanian bioregions – Central Highlands, West, and Southern Ranges (IBRA 7, DCCEEW 2023), in places that have either not burned or burned very infrequently and patchily since the Last Glaciation (Holz et al. 2020). *Subalpine Palaeoendemic Forest and Woodland* covers approximately 325.8 km^2 based on map units – RPF, RKF, RKS, RFS, RPP, RKP and RPW – in TASVEG 4.0 (DPIPWE 2020), representing the most extensive and accurate mapping available within Tasmania for the defined bounds of the ecosystem type at the time of assessment (April 2021).

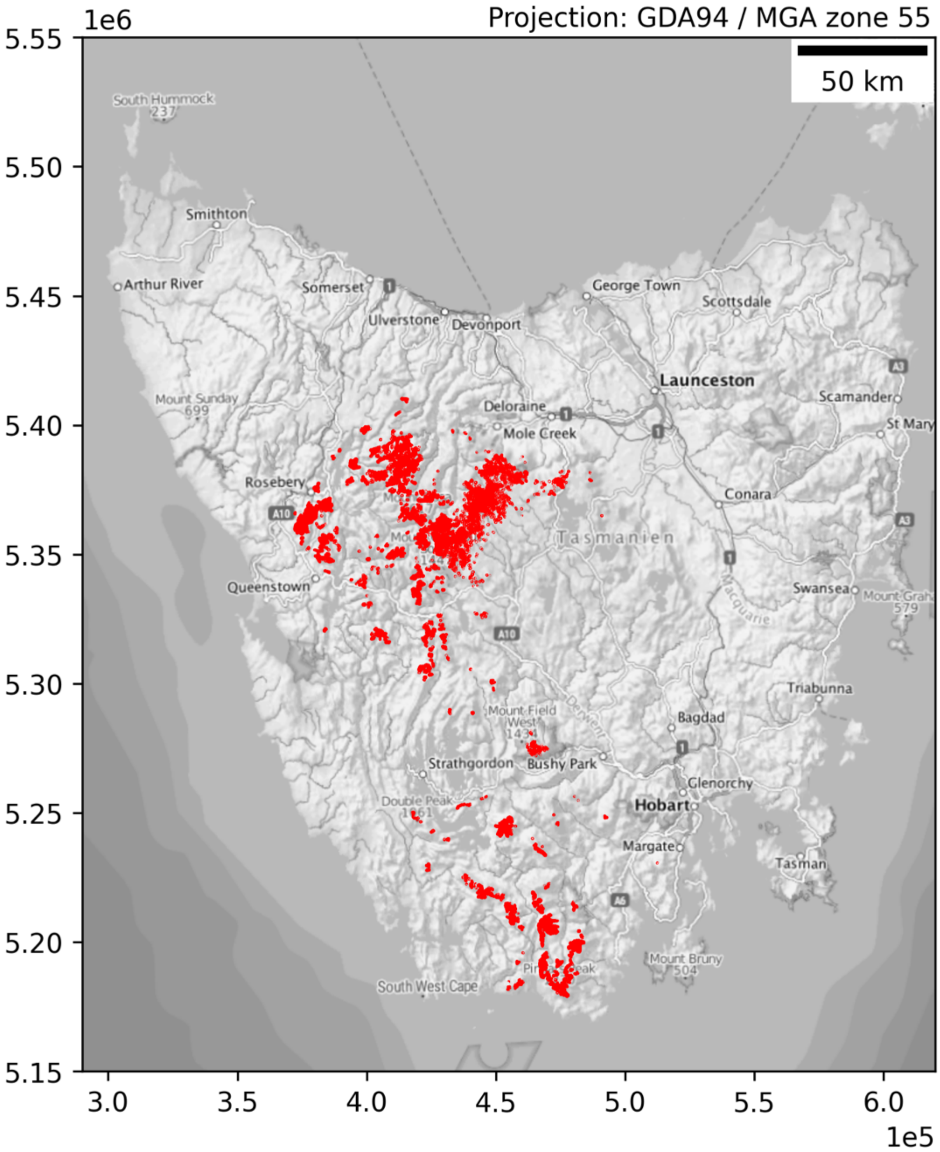


Figure 4. Distribution of *Subalpine Palaeoendemic Forest and Woodland* (red) across Tasmania.

The slow decay and fallrate of fire-killed trees enables stands that were killed several hundred years ago to be identified and mapped (Holz et al. 2020). Kirkpatrick & Dickinson (1984) estimated that 40% of the area of this ecosystem type has been lost to fire since the European invasion. Subsequent fires have further reduced the distribution, most recently in 2016 when a fire burnt 141 ha of the forest at Lake Mackenzie in the Central plateau (Figure 5), killing approximately 69% of the dominant conifers (Bowman et al. 2019; Bliss et al. 2021). In the last 50 years, 150 ha of high altitude (> 600 m) *Athrotaxis*-dominated vegetation has burned, up to 141 ha in the last 20 years (Bliss et al. 2021). Since the European invasion, between 87 km^2 (RKX TASVEG 4.0; (DPIPWE 2020) and 160 km^2 (Brown 1988) of the parts of this community dominated by *Athrotaxis selaginoides* have been destroyed by fire and smelter fumes. Additional areas of *Subalpine Palaeoendemic Forest and Woodland* were lost to fire in the centuries prior to European colonisation (Holz et al. 2020) and earlier in the Holocene (MacPhail 1979, 1981). Modelling of future distributions under projected climate change indicates ongoing loss, even under scenarios of 90% fire suppression (Yospin et al. 2015).

Figure 5. A stand of *Subalpine Palaeoendemic Forest and Woodland* burnt in January 2016 showing partial survival of the tree canopy of *Athrotaxis cupressoides*. Scars from previous fires (c. 1960) can be seen on the trunks of some individuals. There were some post-fire recruits in this area, under browsing pressure from macropods. Lake Mackenzie area, Central Plateau, Tasmania. (Bliss et al. 2021). Photo: David Keith (November 2019).

Abiotic environment

Subalpine Palaeoendemic Forest and Woodland is confined to areas of high rainfall (> 1200 mm p.a.) and mid-high elevations (> 600 m above sea level), where it occurs on a wide range of soil types (including dermosols, podsols, and organosols) derived from a wide range of rock types (quartzites to dolerites; (Kitchener & Harris 2013). The low evapotranspiration rates associated with high elevation (Figure 6), in combination with high precipitation rates, maintain a positive water balance most of the time. Extended dry periods can occur due to variation in the Southern Annular Mode, resulting in intermittently high risk of fire (Yospin et al. 2015; Holz et al. 2020). As the relative absence of fire defines the distribution of the ecosystem type within a high moisture availability envelope, it is more likely to occur on south-facing slopes than north-facing slopes (Venn et al. 2017).

Figure 6. Long unburnt *Subalpine Palaeoendemic Forest and Woodland* on a slope above Tarn Shelf, Mt Field, Tasmania. Photo: Jamie Kirkpatrick.

Characteristic native biota

The diagnostic taxa for the ecosystem type are the winter deciduous *Nothofagus gunnii* and gymnosperms in the genus *Athrotaxis* (Figure 5). These tree species are palaeoendemic, dating from the Cretaceous (Jordan et al. 2016). This ecosystem type also includes palaeoendemic species from other clades, such as the understorey plants – *Diselma archeri*, *Pherosphaera hookeriana* and *Milligania* spp.

There are few mammal and bird species in this ecosystem type (Kirkpatrick et al. 1993). In the more open communities, Bennett's wallaby (*Macropus rufogriseus*) and the Common Wombat (*Vombatus ursinus*) can be found. The most common bird species is the Tasmanian endemic Clinking Currawong (*Strepera versicolor arguta*; a subspecies of Grey Currawong) (Kirkpatrick et al. 1993; Driessen & Mallick 2003). Winter active invertebrates of ancient origin occur within the ecosystem type, such as *Promecoderus* carabid beetles, *Dirce* moths, and *Oreixenica* butterflies (Kirkpatrick et al. 1993; Mallick & Driessen 2005). Some invertebrates are confined to this ecosystem type and related ecosystem types that include the same plant species (Kirkpatrick et al. 1993).

Key components, processes, and interactions

Fire

Subalpine Palaeoendemic Forest and Woodland is typically too wet to burn due to high precipitation, low evapotranspiration, and dense vegetation that maintains a moist microclimate (Jordan et al. 2016) (Figure 7). However, interannual weather variation related to the Southern Annular Mode produces rare extended droughts that render the ecosystem flammable (Holz et al. 2020). Until 2000, the ignition source for most fires burning into this ecosystem type is thought to have been anthropogenic, but more recently, fires started by dry lightning events have become larger and more frequent (Styger et al. 2018).

Recruitment

Seedling recruitment of the dominant palaeoendemic trees is infrequent and occurs in unburnt conditions during high rainfall periods (Holz et al. 2020). The palaeoendemic plant taxa lack regenerative organs and seed banks which, together with their limited dispersal ability, makes them prone to elimination by a single canopy fire. Seedling recruitment is limited and likely during periodically wet years associated with interannual variation in the Southern Annular Mode (Holz et al. 2020). *Athrotaxis*, *Diselma* and *Pherosphaera* may spread clonally by layering. Individual stems of the former genus have been dated at over 1700 years old (Holz et al. 2015).

In the Southern Ranges, recruitment of *Athrotaxis selaginoides* is strongly linked to negative values of the Southern Annular Mode; values have trended positive over the past 300 years, making conditions less suitable for recruitment (Holz et al. 2020). If eliminated from an area by a fire, they may remain absent for centuries or millennia (MacPhail 1979, 1981), unless rare long-distance dispersal occurs. Even where seed sources are close by, post-fire re-establishment can be extremely slow. For example, the density of *Athrotaxis selaginoides* recruits 150 years post-fire was 70 times lower than the pre-fire population at Abrotanella Rise in southern Tasmania (Holz et al. 2020).

Cold and heat-tolerance

The palaeoendemic dominant plants are highly cold-tolerant. The foliage of conifers is reduced to compact imbricate scales with thick cuticles (Read & Hill 1988). *Nothofagus gunnii* is Australia’s only winter-deciduous plant species, enabling its foliage to avoid the most severe winter frosts (Kirkpatrick 1997). Conversely, the same species are susceptible to heat stress and adult plants may suffer dieback or disease when temperatures above 25°C are sustained for several days (Visoiu & Whinam 2015).

Herbivory and snow

The more open parts of the Tasmanian subalpine zone can be well-populated with mammalian herbivores, which consume seedlings of *Athrotaxis cupressoides* and topiarise the adult foliage (Cullen & Kirkpatrick 1988). High populations of mammals are associated with years when snow does not persist. Wallabies die *en masse* when snow is deep and prolonged. Such events may have become rarer since the 1970s, although there are no reliable data and the incidence of snow appears to be cyclic rather than directional (Kirkpatrick et al. 2017). The great longevity of individuals of *A. cupressoides* mitigates against elimination by browsing.

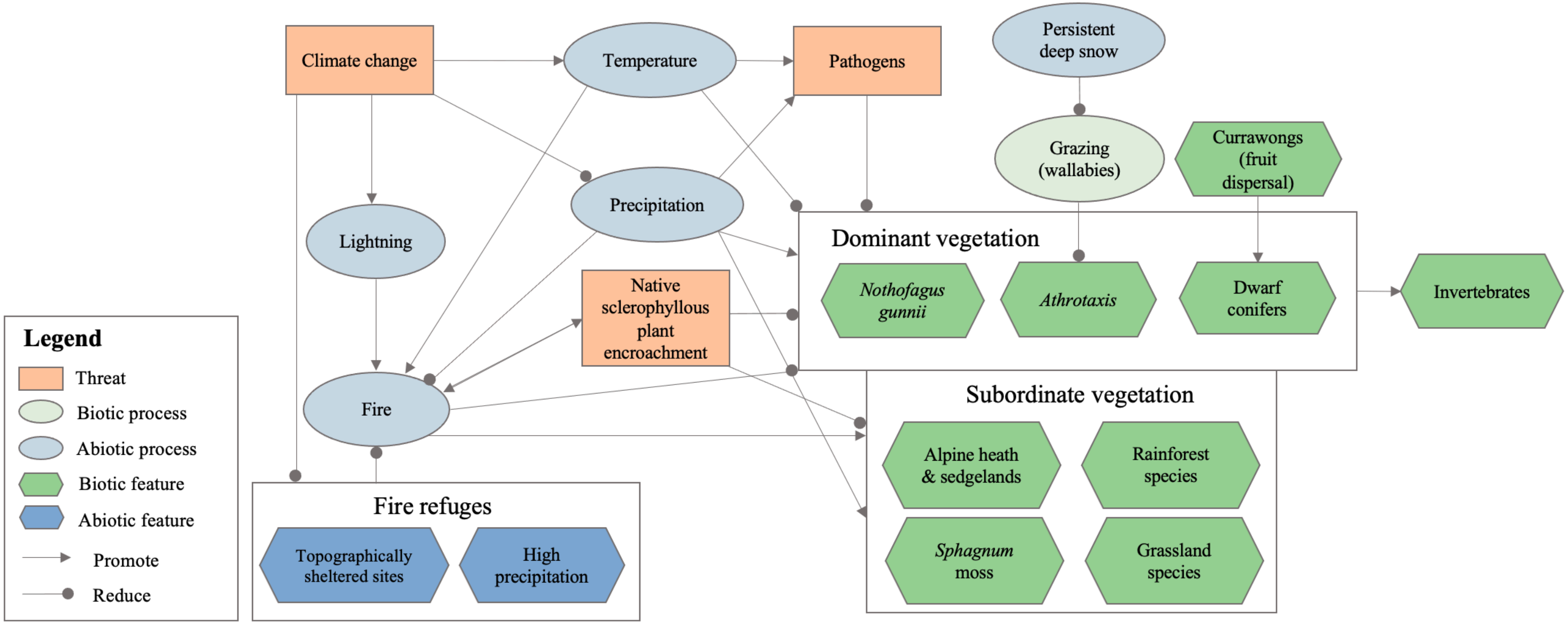


Figure 7. Conceptual diagram illustrating relationships between, and threats to, biotic and abiotic components and processes in *Subalpine Palaeoendemic Forest and Woodland*

Major threats

Fire

The primary threat to *Subalpine Palaeoendemic Forest and Woodland* is the past and future is fire. Complete canopy or lower trunk scorch from a single fire may eliminate the palaeoendemic dominants that define the ecosystem type (Kirkpatrick et al. 2010; Fletcher et al. 2014). Trees that manage to survive but were partially burnt from a fire are also more vulnerable to a second fire, as the fire tends to ignite the dead branch wood. The likelihood of fire has increased substantially since 1970 due to an increase in frequency of drought conditions and dry lightning events. Dry lightning seems to relate to increased variability in temperatures and rainfall during summer and autumn (Styger et al. 2018) and increased atmospheric instability related to a steepening north-south atmospheric pressure gradient (Kirkpatrick et al. 2017). These processes are interactive, systematic, and stochastic. The incidence of a single canopy fire immediately induces collapse. Some evidence suggests that fire can allow the invasion of native sclerophyllous plant species. In 1934 and the 1890s, there were huge losses of Subalpine Palaeoendemic Forest and Woodland in single fires (Kirkpatrick & Bridle 2013). Climate change is expected to result in more frequent and more extensive fires in the future, unless spread of fires from numerous simultaneous lightning strikes can be prevented (Yospin et al. 2015).

Heat and drought stress

Increasing exposure to heat stress and drought have resulted dieback of palaeoendemic trees, which is also likely to accelerate ecosystem decline. Climatic change has been more extreme on the Central Plateau than in the west and is predicted to continue to be so in the future (White et al. 2010). Dieback of *Nothofagus gunnii* has been associated with atypically high temperatures at Marions Lookout, Cradle Mountain (Visoiu & Whinam 2015). Dieback of *Athrotaxis cupressiodes* has been observed on the Central Plateau at Pine Lake, and dieback of *Athrotaxis selaginoides* has been observed near Lake Fenton on Mt Field. These localities are at the dry edges of the ranges of the two species.

Macropods

High densities of macropod browsers may also threaten characteristic elements of the ecosystem type, but these effects are likely to be localised and at least somewhat transient.

Introduced species and pathogens

This ecosystem type is largely restricted to natural landscapes within relatively remote areas. These areas are at lower risk of invasion by introduced plants and pathogens and the closed nature of the vegetation and its occurrence at high elevations reduce the likelihood of impacts on susceptible components of the vegetation from the most widely distributed plant pathogen, *Phytophthora cinnamomi,* due to low soil temperatures. However, canopy dieback, which may be associated with both native and introduced plant pathogens, have been reported for the dominant trees where climatic stresses have occurred such as heat and/or drought events, including a major dieback event at Pine Lake, Tasmania.

IUCN Stresses Classification

1.2 – Ecosystem degradation
1.3 – Indirect ecosystem effects

IUCN Threats Classification

7.1.1 – Increase in fire frequency / intensity
11.1 – Habitat shifting and alteration
11.2 – Droughts
11.3 – Temperature Extremes

Ecosystem collapse

Ecosystem collapse for *Subalpine Palaeoendemic Forest and Woodland* usually involves fire-driven transition to another ecosystem type, including: *Tasmanian Alpine Heath* dominated by *Richea scoparia;* short rainforest dominated by *Nothofagus cunninghamii* and *Eucryphia milliganii*; or a Eucalypt or Leptospermum dominated forest or scrub. These alternate ecosystem types are dominated by plants that may resprout after fire in some situations. The ecosystem type may also transition into another type if sclerophyll species invade large canopy gaps created by drought/heat caused death of rainforest species. Lastly, the ecosystem type may transition into a low subalpine scrub or heathland ecosystem dominated by *Leptospermum nitidum* that still retains some alpine and rainforest species (e.g., *Athrotaxis* stags)

Subalpine Palaeoendemic Forest and Woodland collapses when any of the following occur:

1) *Area:* The mapped distribution declines to zero (100% loss) (Criteria A and B),
2) *Fire occurrence:* All patches have been burnt since the industrial era (Criterion C), and/or
3) *Tree and shrub cover*: Palaeoendemic trees and shrubs are reduced to densities of < 2 trees per hectare (Criterion D).

Assessment information

Criterion A

Summary

The status of *Subalpine Palaeoendemic Forest and Woodland* under criterion A is **Vulnerable** due to historical distribution declines from fires (sub-criterion A3). Assessments of sub-criteria A1 and A2 produced outcomes of **Least Concern** and **Vulnerable** respectively.

Assessment outcome

Recent change (A1)

Decline in the distribution of *Subalpine Palaeoendemic Forest and Woodland* over the past 50 years (i.e. since 1971) is likely to involve relatively small areas. Historical records of bushfires and planned burns are available as shapefiles for Tasmania (Tasmanian Government 2022). The fire history dataset includes fires in the 1960/1961 and 2019/2020 fire seasons in regions > 600 m above sea level (Tasmanian Government 2022). Fire seasons are defined as fires that occur between July in one year and June in the next year. Based on the spatial layers of fire histories, Palaeoendemic forests have burnt in 15 fire seasons between 1970 and 2020, covering a total of 1.49 km^2 (of 325.8 km^2), or 0.46% of the whole ecosystem type. Subsequent fires have further reduced the distribution, most recently in 2016 when a fire burnt 141 ha of the forest at Lake Mackenzie in the Central plateau (Figure 1), killing approximately 69% of the dominant conifers (Bowman et al. 2019; Bliss et al. 2021). In the last 50 years, 150 ha of high altitude (> 600 m) *Athrotaxis*-dominated vegetation has burned, up to 141 ha in the last 20 years (Bliss et al. 2021). An additional area of up to 1% may have been lost due to inundation during construction of dams in the Central Highlands and West and recreational infrastructure in the 1960s-1970s. In combination, these losses form fire and inundation account for substantially less than 30% of the distribution over the past 50 years. The status of the ecosystem type under sub-criterion A1 is **Least Concern**.

Future change (A2)

Future declines in the distribution of *Subalpine Palaeoendemic Forest and Woodland* depend on the incidence of canopy fires. Spread of such fires is driven by several factors: the probability of ignitions from dry lightning strikes and human sources; the probability of extended dry spells that reduce fuel moisture content prior to ignition; and the probability of severe fire weather events (high wind speeds, high temperatures and low humidity). These scenarios have occurred sporadically in recent centuries. Dendrochronical studies on Abrotanella Rise in the Southern Ranges document a peak in fire-related mortality of *Athrotaxis cupressoides* (Holz et al. 2020). In the summer of 1960-61, ~10% of the ecosystem distribution was burnt a single fire of the western side of the Central plateau (Holz et al. 2015). These events indicate a strong likelihood that considerable areas of *Subalpine Palaeoendemic Forest and Woodland* could be burnt as the climate warms further in the coming decades, as periods on low fuel moisture and extreme fire weather become more frequent.

Styger et al. (2018) found that the mean area burnt by lightning-ignited fires increased during 2000-2016, after a period from 1980 to 2000 when very small areas were burnt by lightning-ignited fires. They were unable to attribute the cause of this change due to insufficient data, but likely explanations include an increase in lightning frequency or an increase in fire spread from lightning strikes.

Climate models project further slight increases in dry lightning and stronger tendencies towards drier summers (Love et al. 2016; Styger et al. 2018; Abram et al. 2021). A statistical model of these factors projected fire regimes and ecosystem dynamics in the Cradle Mountain-Lake St Clair National Park (Yospin et al. 2015). This study showed that the extent of *Subalpine Palaeoendemic Forest and Woodland* contracted greatly under all fire scenarios, including one in which 90% of ignitions were suppressed. Fires in 2025 further demonstrate this issue - with more paleoendemic rainforests being burnt in the Cradle Mountain - Lake St Clair National Park.

There are no available projections for the next 50 years. However, evidence of multiple extensive fires within the distribution of the ecosystem type during the 20th century, trends, and projections of increasing frequency of severe droughts and extreme fire weather, and projections of ongoing decline together suggest that a future reduction in distribution of at least 30% is plausible. Therefore, the status of *Subalpine Palaeoendemic Forest and Woodland* under sub-criterion A2 is **Vulnerable**.

Historical change (A3)

The post-1750 industrial era reached Tasmania soon after 1803 when European people invaded and settled on the island. Historical declines in the distribution and function of the ecosystem type are therefore estimated since that time. The historical distribution of *Subalpine Palaeoendemic Forest and Woodland* around that time is unknown, but very likely included some areas that are currently mapped as *Tasmanian Alpine Heath* (map units HHE and HHW in TASVEG 4.0) and *Tasmanian Alpine Sedgeland* (map units HSE, HSW in TASVEG 4.0), also subalpine heathland and scrub (SSW, SHS) and subalpine rainforest (RSH). Hence, the historical extent *Subalpine Palaeoendemic Forest and Woodland* may be estimated from the current extent of the distribution, plus that of the ecosystem types that are likely to have replaced it after local collapse. The latter include highland rainforest with burnt and killed *Athrotaxis selaginoides* (map unit RKX in TASVEG 4.0), *Tasmanian Alpine Heath* and subalpine occurrences (above 600 m elevation) of *Nothofagus-Atherosperma* rainforest (map unit RMT in TASVEG 4.0).

Based on the most recent mapping (TASVEG 4.0) (DPIPWE 2020) the area of highland rainforest with burnt and killed *Athrotaxis selaginoides* (map unit RKX) is 87 km^2, the area of live *Athrotaxis selaginoides* rainforest (map unit RKP) is 174 km^2, and combined extent of other areas dominated by *Athrotaxis selaginoides* is 83 km^2. The decline in extent of *Athrotaxis* rainforest maybe as low as 25% (100×(1-87.09/(87.09+173.91+83.14)) or as high as 33% (100×(1-87.09/(87.09+173.91)); the higher estimate is more likely. *Subalpine Palaeoendemic Forest and Woodland* also includes communities dominated by other palaeoendemic species (TASVEG 4.0 map units RPF, RPW, RPP, RFS), however, the fire-killed extent of these units is not mapped. Assuming that these other types of forests underwent declines in distribution of a similar magnitude to *Athrotaxis selaginoides* rainforest suggests a lower bound of decline for the whole ecosystem type in the order of 25-33% since 1750.

Much of the current extent of *Tasmanian Alpine Heath* (map units HHE, HHW) with a combined area of 893.79 km^2 may formerly have been either *Subalpine Palaeoendemic Forest and Woodland* or *Tasmanian Coniferous Heath* transformed by major fires. Further, *Subalpine Palaeoendemic Forest and Woodland* may have been transformed into rainforests dominated by resprouters such as *Nothofagus cunninghamii* and *Eucryphia milliganii* (subalpine areas within map unit RMT) (Kirkpatrick 1997). Although, the timing of transformation events is not known precisely, much of the historical collapse may have occurred after 1803. Around this time, there was increase in the frequency of large fires recorded in the dendrochronological record, and increased severity of fire weather conditions represented by the Southern Annular Mode (Holz et al. 2020). This inference is further supported by records of extensive fires in central, western, and southern Tasmania during the 1890s (mainly 1897-98), 1930s (mainly 1933-34), 1950s, and 1970s (Marsden-Smedley 1998). The early prospecting and surveying expeditions during the 19th century are also reported to have burnt large areas of the landscape but later fires prevented mapping the earlier fires (Marsden-Smedley 1998).

A plausible estimate of the area of *Subalpine Palaeoendemic Forest and Woodland* transformed into heathlands and other forms of rainforest and scrub may be derived by assuming that half of the currently mapped extent of *Tasmanian Alpine Heath* (446.90 km^2) had been part of the *Subalpine Palaeoendemic Forest and Woodland* ecosystem type until sometime after 1803, in addition to the area of mapped fire-killed *Athrotaxis selaginoides* (87 km^2). This enables an alternative estimate of decline in distribution since 1750 of 67.5% (100×(446.9+87.09)/(446.9+87.09 +173.91+83.14)). A smaller area of *Nothofagus-Atherosperma* rainforest is likely to be derived from burnt *Subalpine Palaeoendemic Forest and Woodland*. We assumed 10% of the extent of the subalpine area (above 600 m elevation) of *Nothofagus-Atherosperma* rainforest (i.e. 10% of 132 km^2) was *Subalpine Palaeoendemic Forest and Woodland* prior to 1803, producing an upper bound of overall estimated decline of 72.2% (100×(446.9+87.09+131.96)/(446.9 +87.09+173.91+83.14+131.96)). The estimate could be slightly higher (1-2%), given that it excludes areas of the ecosystem type eliminated by the 2016 fire and development of infrastructure including dams and recreational facilities.

Based on these scenarios, the plausible estimates of decline in distribution of *Subalpine Palaeoendemic Forest and Woodland* since *1750* span a range of 25–74% (median 49%). The status of the ecosystem type under sub-criterion A3 is therefore **Near Threatened (Least Concern – Endangered)**.

Criterion B

Summary

The status of *Subalpine Palaeoendemic Forest and Woodland* under criterion B is **Vulnerable (Vulnerable – Endangered),** based on the extent of occurrence (EOO; sub-criterion B1).

Assessment Outcome

Sub-criterion B1

The extent of occurrence (EOO) was determined using existing map products from Tasmania (Kitchener & Harris 2013; DPIPWE 2020). The minimum convex polygon enclosing all mapped occurrences of *Subalpine Palaeoendemic Forest and Woodland* has an area of 20,786.13 km^2 (Figure 8). This area is within the threshold values for **Vulnerable**, close to the threshold for Endangered. Based on the ongoing transformation of the ecosystem type by fires, there is a continuing decline in the distribution and function of the ecosystem type and the ongoing incidence of subalpine fire is a threat likely to cause ongoing declines. The ongoing incidence of fires is causing continuing declines and fragmentation in the distribution and function of the ecosystem type. Given historical and recent precedents (Abram et al. 2021), the entire distribution of the ecosystem type could be burnt in a small number of extensive fires over the next 20 years. It is therefore inferred to occupy 1-5 threat-defined locations (see Sub-criterion B3). *Subalpine Palaeoendemic Forest and Woodland* therefore meets all three sub-criteria within B1 for listing as **Vulnerable (Vulnerable – Endangered)**.

Sub-criterion B2

The area of occupancy (AOO) was determined using the same map as used in sub-criterion B1. The distribution of *Subalpine Palaeoendemic Forest and Woodland* intersects with a minimum of 112 10 × 10 km grid cells (Figure 8). The status of the ecosystem type under sub-criterion B2 is therefore **Least Concern**.

Sub-criterion B3

The number of threat-defined locations was based on fire, as this is the most important stochastic threat to *Subalpine Palaeoendemic Forest and Woodland*. The ecosystem type is estimated to occupy 1-5 threat-defined locations as historical and recent fires (Abram et al. 2021) suggests that the entire distribution could be burnt in a small number of extensive fires, causing the ecosystem type to collapse or become Critically Endangered within a very short period (c. 20 years). It therefore meets the requirements for **Vulnerable** status under sub-criterion B3.

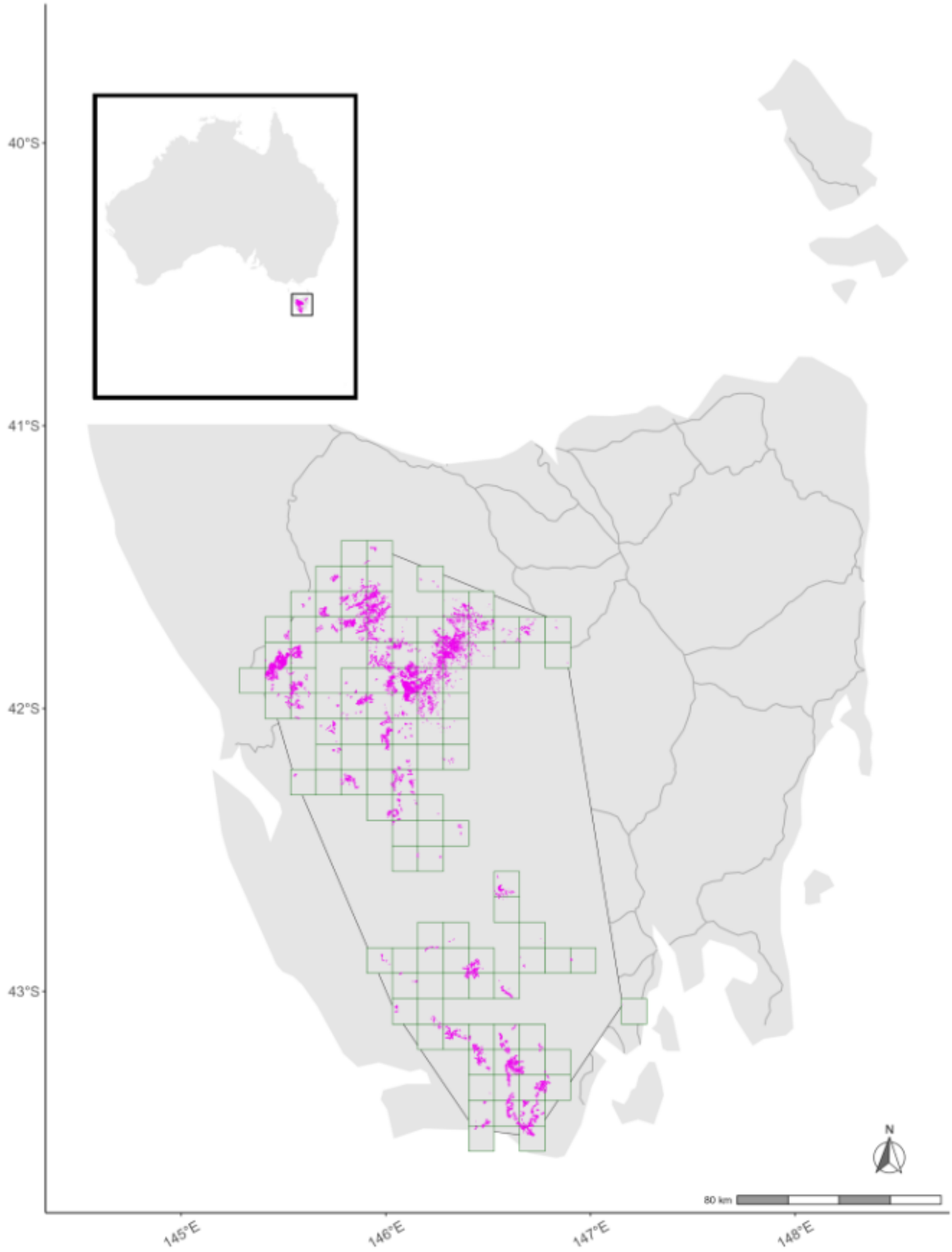


Figure 8. Map of *Subalpine Palaeoendemic Forest and Woodland* (magenta polygons) across Australia, showing Extent of Occurrence (black polygon) and Area of Occupancy, where the 1% rule was not applied (green squares).

Criterion C

Summary

Fires are the major source of environmental degradation of *Subalpine Palaeoendemic Forest and Woodland* and cause mortality of the characteristic dominant palaeoendemic trees. In this assessment, the effect of fire is addressed under criterion A through estimates of the area in which dominant palaeoendemic trees have been eliminated. Other fire effects that result in altered ecosystem structure or reduced density of palaeoendemic trees without eliminating them at stand-scale are poorly known and hence there are insufficient data to assess them under criterion C. Environmental degradation may also be caused by climate change, which is projected to result in increased frequency of heat waves and droughts associated with dieback of palaeoendemic trees. Preliminary monitoring of conifer canopy condition over time suggested a slight declining trend over the last decade (Visoiu & Balmer 2024). However, at present, insufficient data are available to reliably assess this process. The status of the ecosystem type is **Data Deficient** under criterion C.

Identification of indicators

We examined the relevance and data availability/quality for two abiotic indicators, selected based on our conceptual model, to assess the risk of collapse from environmental degradation:

- Fire occurrence: a direct measure of occurrence of fires in Subalpine Palaeoendemic Forest and Woodland.

Indicator: Fire occurrence

Relevance to ecosystem function

Subalpine Palaeoendemic Forest and Woodland is closely associated with sheltered, long-unburnt sites (Venn et al. 2017) and are driven to collapse by canopy fires (Holz et al. 2015, 2020). The likelihood of fire in Tasmania has considerably increased since 1970 due to more frequent ignitions (Kirkpatrick et al. 2017; Styger et al. 2018). Palaeoendemic trees that dominate this ecosystem type lack seed banks and the capacity to recover vegetatively from fire, and have limited dispersal ability (Kirkpatrick & Bridle 2013; Fletcher et al. 2014), so complete scorching of the canopy from a single fire can eliminate the characteristic palaeoendemic species (Kirkpatrick et al. 2010). Therefore, *Subalpine Palaeoendemic Forest and Woodland* is fire-sensitive, as a single fire event can cause the ecosystem to transform into more fire-resilient ecosystem types (Fletcher et al. 2014).

Selection of collapse threshold

A single fire can cause local collapse of *Subalpine Palaeoendemic Forest and Woodland* for centuries (Kirkpatrick et al. 2010) as most plant species cannot recover vegetatively after fire damage and have limited dispersal ability (Kirkpatrick & Bridle 2013). Therefore, we consider the ecosystem type to collapse when all patches have been burnt since the industrial era.

Data availability and quality

A suitable analysis requires time series data and projections for ignition frequencies, fuel moisture (e.g., Forest Fire Danger Index or soil moisture) and incidence of extreme fire weather (high temperatures, low humidity, strong winds). These data exist but have not been analysed or modelled to estimate the incidence or probability of fire across the distribution of the ecosystem. Therefore, we could not adequately assess this indicator. The impact of fires is indirectly assessed under Criterion A.

Criterion D

Summary

Heatwaves and droughts disrupt biotic processes in *Subalpine Palaeoendemic Forest and Woodland* by causing tree mortality and declines in tree density. Insufficient data are available to assess this process, and hence the status of the ecosystem type is **Data Deficient** under criterion D.

Identification of indicators

We examined the relevance and data availability/quality for one indicator to assess the risk of collapse from disruption of biotic processes and/or interactions:

- *Tree and shrub cover*: a direct measure of the cover of the characteristic trees and shrubs in *Subalpine Palaeoendemic Forest and Woodland*

Indicator: Tree and shrub cover

Relevance to ecosystem function

Subalpine Palaeoendemic Forest and Woodland is characterised by palaeoendemic tree and shrub species. In particular, the tree species *Nothofagus gunnii* and gymnosperms in the genus *Athrotaxis*, and understorey plants *Diselma archeri*, *Pherosphaera hookeriana, Milligania spp.* (Kitchener & Harris 2013). The loss of these palaeoendemic tree and shrub species would lead to the loss of the ecosystem type and transition *to Tasmanian Alpine Heath* (dominated by *Richea scoparia*) or short rainforest (dominated by *Nothofagus cunninghamii* and *Eucryphia milliganii*).

Selection of collapse threshold

The ecosystem type is considered to collapse when the cover of Palaeoendemic tree and shrub species is < 1-2 trees per hectare.

Data availability and quality

While some timeseries estimates of the density of palaeoendemic trees have been constructed from counts of live and dead trees (Holz et al. 2020), insufficient data are currently available to assess this process, hence we could not adequately assess this indicator. A monitoring program to track the condition of conifers has been established in the Tasmanian Wilderness World Heritage Area was established in 2011 (Fitzgerald 2011) and some preliminary results for the first decade of the monitoring have been reported (Visoiu & Balmer 2024) but only indicate a slight downward trend in condition to-date.

Criterion E

Summary

Insufficient data are available to develop a model to estimate the probability that *Subalpine Palaeoendemic Forest and Woodland* will collapse within the next 50 to 100 years. Hence the ecosystem type is **Data Deficient** under criterion E.

Athrotaxis cupressoides and *Nothofagus gunnii,* Cradle Mountain-Lake St Clair National Park, Tasmania. Photo: Peter Dombrovskis (1986).

Alpine-subalpine Woodland and Forest

Authors

Chloe Sato, Jessica Rowland, Ella Plumanns Pouton, Kate Umbers

Reviewers

Emily Nicholson, Matt White

Biome

T4 Savannas and grasslands

Functional group

T4.4 Temperate woodlands

IUCN status

Near Threatened

Perisher Ski Resort in winter, Kosciuszko National Park. Photo: Lois Padgham.

Assessment Summary

Alpine-subalpine Woodland and Forest is dominated by a low-statured *Eucalyptus* trees, with an understorey of grasses, herbs, and shrubs. The ecosystem type is sensitive to drought stress, grazing by exotic herbivores, and increasing frequency of fires. The status of the ecosystem type is assessed as **Near Threatened** due to an increase in the frequency of fires over the last 50 years (Table 6).

Table 6. Summary of the Red List of Ecosystems assessment of the *Alpine-subalpine Woodland and Forest.* Category ranges in parentheses reflect uncertainty in assessment under the corresponding criteria.

Criteria	A	B	C	D	E	Overall
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	LC	NT	DD	DD	NT
Sub-criterion 2 *A, C, D: 50-year period including present & future* *B: AOO*	LC	LC	LC	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	LC	LC	DD	DD		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria

Ecosystem description

Ecosystem classification

In the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022a), *Alpine-subalpine Woodland and Forest* belongs to the Ecosystem Functional Group *T4.4 Temperate Woodlands*, within the *Savannas & Grasslands* biome.

Under national classifications, *Alpine-subalpine Woodland and Forest* falls under Subalpine Woodlands Major Vegetation Subgroup (Keith & Pellow 2015). Under regional classifications, woodland and forest characteristic of the subalpine zone falls under Scabby Range dry shrub woodland (Vegetation Group 37), Western subalpine moist shrub forest (Vegetation Group 86), Western montane moist shrub forest (Vegetation Group 98), Montane dry shrub/herb/grass forest (Vegetation Group 99), Subalpine dry shrub/herb/grass woodland (Vegetation Group 127), Subalpine dry shrub/herb woodland (Vegetation Group 128), Subalpine shrub/grass woodland (Vegetation Group 130) in the Australian Capital Territory ('ACT') and in New South Wales ('NSW') (Gellie 2005); Subalpine woodland in Victoria (Ecological Vegetation Class (EVC) 43) (DSE 2004); *Eucalyptus coccifera* forest and woodland ("DCO") and *Eucalyptus gunnii* woodland ("DGW") in Tasmania (Kitchener & Harris 2013).

Distinction from similar ecosystem types

Alpine-subalpine Woodland and Forest is structurally similar to *Subalpine Palaeoendemic Forest and Woodland,* but is distinguishable by its characteristic plant species. *Alpine-subalpine Woodland and Forest* is dominated by low trees (< 12 m) such as *Eucalyptus pauciflora* subsp. *niphophila* and *Eucalyptus pauciflora* subsp. *pauciflora* on the mainland (Venn et al. 2017) or *E. coccifera*, *E. gunnii*, and *E. subcrenulata* in Tasmania (Kitchener & Harris 2013; Venn et al. 2017). There is an understorey of grasses, herbs, and shrubs. In contrast, *Subalpine Palaeoendemic Forests and Woodland* is defined by the winter deciduous *Nothofagus gunnii* and gymnosperms in the genus *Athrotaxis*, which date back to the Cretaceous (Jordan et al. 2016), and several other clades, including the understorey plants, *Diselma*.

Distribution

Australian *Alpine-subalpine Woodland and Forest* occurs in the mountainous environments of south-eastern Australia (ACT, NSW, Victoria and Tasmania). The ecosystem type is located between 145.68° to 149.11° longitude and between -35.35° and -43.49° latitude. The altitudinal band occupied differs between states and territories in Australia (Costin & Wimbush 1972). It is generally found at 800-1200 m ASL in Tasmania (i.e., above the upper limit of *Eucalyptus delegatensis* dominated communities), 1350-1900 m in ACT, 1350-1750 m in Victoria, and 1400-1830 m in NSW.

In total, *Australian Alpine-subalpine Woodland and Forest* covers an area of approximately 4,435.0 km^2 (Figure 9). Spatial products used to create this map represent the most extensive and/or accurate mapping available within each state for the defined bounds of the ecosystem type at the time of assessment (April 2021), as advised by representatives from each government department involved in the assessment process.

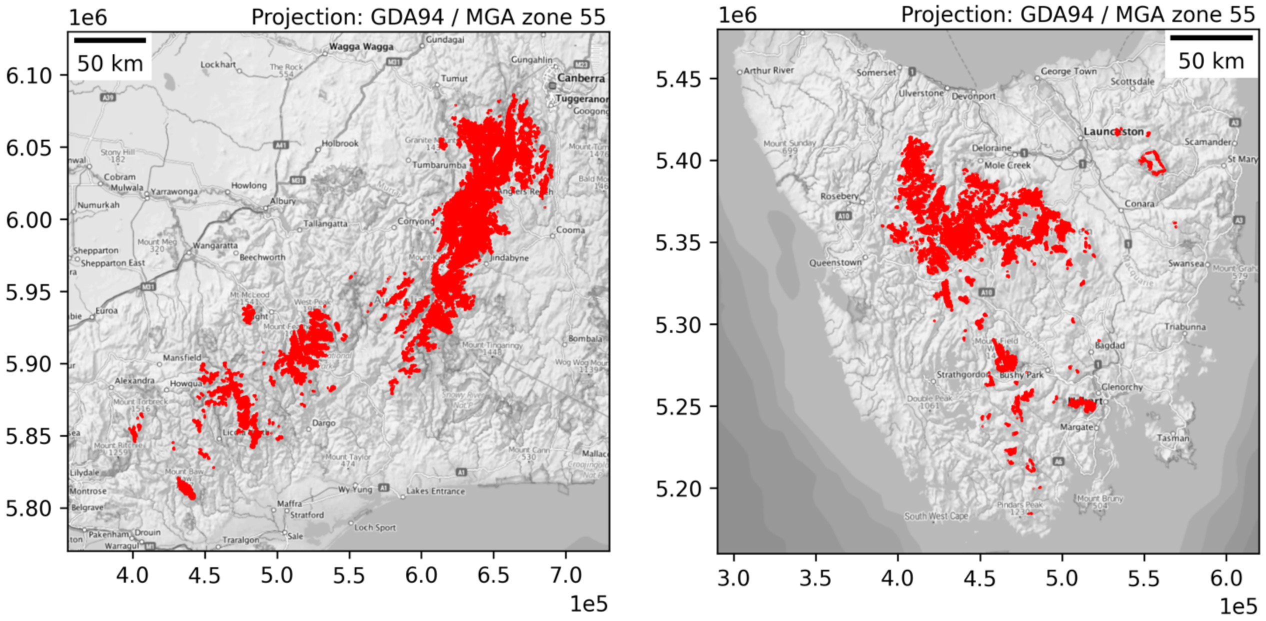


Figure 9. Distribution of Al*pine-subalpine Woodland and Forest* (red) in mainland Australia (left) and across Tasmania (right).

Abiotic environment

Alpine-subalpine Woodland and Forest is characterised by low mean mid-summer temperatures (maximum between 15°C to 24°C), cold mid-winter temperatures (minimum between -4°C to -1°C) (Costin 1957), and mean temperatures during the growing season between 6.5°C and 10°C (Green & Stein 2015). For about six months of the year, minimum temperatures are below freezing (Green & Osborne 2012; Green & Stein 2015) and severe frosts are common during snow-free months (Costin 1954). Mean annual precipitation is high (approximately 760-2030 mm) (Costin 1957), and up to 60% of this precipitation falls as snow (Green & Osborne 2012) covering the ground for between one to four months per year (Costin 1957; Green & Osborne 2012).

On the mainland, the underlying soils are highly erodible (Gibbons & Rowan 1993), acidic to strongly acidic, and low in available nutrients (Costin 1954).The soils exhibit high water holding capacity and rapid rates of organic matter decomposition (Costin 1954). In Tasmania, *Alpine-subalpine Woodland and Forest* is associated with well-drained dolerite substrates (Williams & Potts 1996; Reid & Potts 1999).

Characteristic native biota

Alpine-subalpine Woodland and Forest is dominated by low trees (usually < 12 m) (Figure 10). On mainland Australia, *Eucalyptus pauciflora* subsp. *niphophila* and *Eucalyptus pauciflora* subsp. *pauciflora* are the characteristic subalpine woodland tree species, but other *Eucalyptus* subspecies (*E. pauciflora* subsp. *acerina*, *debeuzevillei*, *hedraia*) and species (*E. lacrimans*, *E. stellulata*) may be locally dominant (Venn et al. 2017). In Tasmania, *E. coccifera*, *E. gunnii*, and *E. subcrenulata* are the characteristic tree species (Kitchener & Harris 2013; Venn et al. 2017).

The understorey is discontinuous and dominated by grasses (*Poa* spp., *Rytidosperma* spp.), herbs (*Celmisia* spp., *Ranunculus* spp.), and/or shrubs (e.g., *Hovea* spp., *Oxylobium* spp., *Orites* spp., *Bossiaea* spp., *Grevillea* spp., *Ozothamnus* spp., *Leptospermum* spp., *Hakea* spp., *Richea* spp., *Leucopogon* spp., *Epacris* spp.). The type of understorey is dependent on soils (deep versus rocky, parent material) and the exposure to fire (Costin 1957). Often, the understorey is floristically indistinguishable from adjacent treeless vegetation communities (Venn et al. 2017). Around 211 species and subspecies of plant occur in association with *Alpine-subalpine Woodland and Forest* across NSW (Costin 1954). Armstrong et al. (2013) recorded a mean richness of 24-28 (± 6 standard deviation) plant species in 0.04 to 0.1ha plots in *Eucalyptus pauciflora* subsp. *debeuzevillei* and *Eucalyptus pauciflora* subsp. *niphophila* dominated woodland across NSW and the ACT (higher elevation woodlands), which is roughly equivalent to the plot scale plant richness described for *Eucalyptus pauciflora* dominated woodland in Victoria (~ 27 species) (DSE 2004). In Tasmania, condition monitoring benchmarks indicate that plant richness varies between 21 species (*E. gunnii* forest and woodland) and 38 species (*E. pauciflora* subsp. *pauciflora* grassy woodland and forest) (DPIPWE 2016a).

The vegetation of *Alpine-subalpine Woodland and Forest* provides diverse niches that fauna can occupy. In terms of richness, 24 native mammal species, > 45 bird species, and 11 reptile species have been recorded in the ecosystem type, as well as a variety of invertebrates - many of which are not found in other alpine-subalpine ecosystems (Green & Osborne 2012) and/or are threatened (e.g., Key's matchstick grasshopper *Keyacris scurra*). Possums (e.g., common ringtail possum *Pseudocheirus peregrinus*), bats (e.g., Gould's wattled bat *Chalinolobus gouldii*), and birds (e.g., crimson rosella *Platycercus elegans*, white-throated treecreeper *Cormbates leucophaea*, striated pardalote *Pardalotus striatus*) use tree hollows in live mature trees. Small mammals (e.g., broad-toothed rat *Mastocomys fuscus*, southern bush rat *Rattus fuscipes*, agile antechinus *Antechinus agilis*), foraging and breeding birds (e.g., flame robin *Petroica phoenicea*, olive whistler *Pachycephala pectoralis*, white-browed scrub-wren *Sericornis frontalis*, white-eared honeyeater *Lichenostomus leucotis*), and reptiles (e.g., Guthega skink *Liopholis guthega*, metallic skink *Niveoscincus metallicus*, white-lipped snake *Drysdalia coronoides*) use the complex understories of shrub-dominated woodland and forest. *Alpine-subalpine Woodland and Forest* near water sources often supports frogs (e.g., northern corroboree frog *Pseudophryne pengilleyi*, southern toadlet *Pseudophryne dendyi*, baw baw frog *Philoria frosti*) outside the breeding season, as well as invertebrates (e.g., Riek's crayfish *Euastacus rieki*, alpine stonefly *Thaumatoperla alpina*). Threatened grasshoppers such as the brown skyhopper *Kosciuscola* cuneatus, lightening skyhopper *K. usitatus*, Mt. Buffalo skyhopper *K. restrictus* and thermocolour skyhopper *K. tristis* also occur in *Alpine-subalpine Woodland and Forest*, along with other grasshopper species such as the montane skyhopper *K. cognatus*. Invertebrates such as longicorn beetles (*Phorocantha* spp.) and wingless soldier fly (*Boreoides subulatus*) use the trees as a primary resource for completing their lifecycle, and the endangered Bogong moth *Agrostis infusa* feeds on flowers throughout *Alpine-subalpine Woodland and Forest*.

Figure 10. *Alpine-subalpine Woodland and Forests* in Kosciusko National Park, 2011. Photo: Chloe Sato.

Key components, processes, and interactions

Dieback

Snowmelt supplies subalpine soils with water that eucalypts access during spring and summer to support their growth and persistence (Figure 11). In poor snow years and years of drought, eucalypts are unable to alter their evapotranspiration potential, becoming stressed during summer months. During extended periods of drought, hydraulic failure may lead to canopy death (e.g., as seen in *E. pauciflora* at slightly lower elevations) (Nolan et al. 2021). This stress can also increase the susceptibility of individual trees to insect attack (e.g., wood-boring longicorn beetles, *Phorocantha* spp.) (Seaton et al. 2015). If the duration and severity of drought is lengthy, it may lead to the death of stressed and/or insect-affected trees. Dieback is a natural process, but under changing climatic conditions, it is now considered an emerging threat to *Alpine-subalpine Woodland and Forests* (Clarke 2021).

Fire

Fire was likely an infrequent disturbance in *Alpine-subalpine Woodland and Forest* prior to European colonisation (e.g., fire-free intervals of more than 100 years in some areas) (Good 1982; Zylstra 2006) (Figure 11). In post-colonisation conditions, fire can kill mature trees (depending on its severity) (Coates et al. 2012). A flush of regeneration may occur post-fire, but if fire return intervals are short, death of regeneration and new recruits will occur. Increased fire frequency increases tree mortality (Fairman et al. 2017), reduces seedling recruitment (Fairman et al. 2017), decreases available hollows in characteristic tree species (e.g., *E. pauciflora*) (Salmona et al. 2018), and shifts shrubby understories to those dominated by grasses (Fairman et al. 2017). Further, insect-related dieback events are more prominent after fire, as longicorn beetles likely target remaining live trees in post-fire environments.

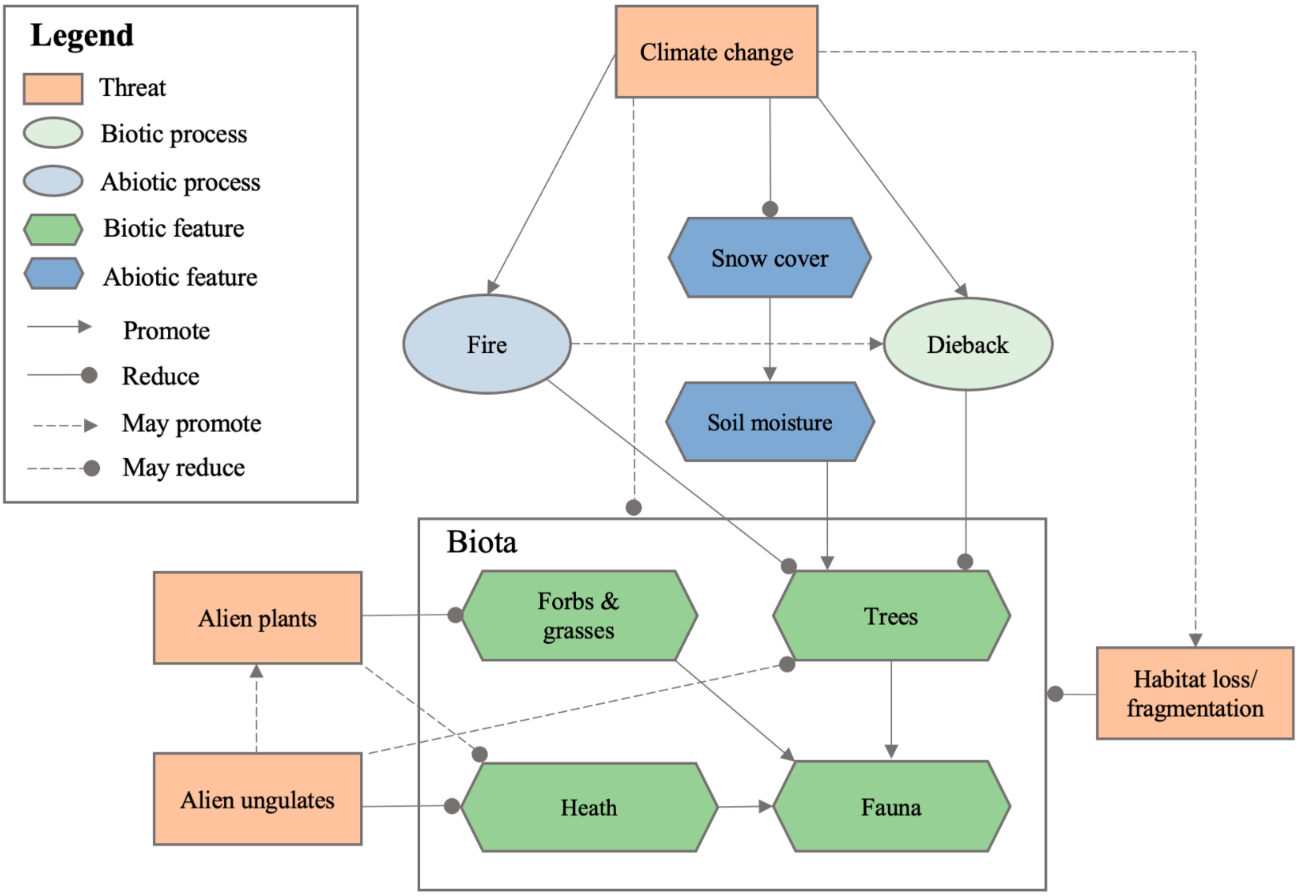


Figure 11. Conceptual diagram illustrating relationships between, and threats to, biotic and abiotic components and processes in *Alpine-subalpine Woodland and Forest*.

Major threats

Threats to *Alpine-subalpine Woodland and Forest* include dieback, fire, weed invasion, grazing by feral herbivores, habitat loss and fragmentation from infrastructure development, and climate change (Figure 11).

Dieback

While a natural process, the declining health and death of snow-gums (particularly *E. pauciflora* subsp. *niphophila*, *E. pauciflora* subsp. *debeuzevillei*; Brookhouse, pers. comm) is an emerging threat to *Alpine-subalpine Woodland and Forest*. Historical records of high-elevation dieback events are scarce, but Shields (1993) documented progressively expanding dieback outbreaks from aerial photography taken during 1964, 1978 and 1988 in NSW. Recent observations indicate that snow-gum dieback is widespread throughout the Australian Alps (SOSnowgum 2023) with national parks exhibiting signs of dieback in NSW (Kosciuszko National Park), ACT (Namadgi National Park), and Victoria (Alpine National Park). The dieback appears to be most prevalent between 1600 m to 1800 m elevation but can affect some lower elevation eucalypt species (e.g., *E. lacrimans*).

Snow-gum dieback is, in part, attributed to outbreaks of wood-boring longicorn beetles from the *Phoracantha* genus (Clarke 2021). Beetles in this genus feed within the wood and bark of host trees, and severe infestations result in ringbarking of trees. Ringbarking severely impairs the flow of water and carbohydrates around the tree, leading to the decline and death of the tree. Outbreaks of *Phoracantha* beetles in living trees are generally a product of drought stress (e.g., Seaton et al. 2015). With increased frequency, intensity, and duration of drought conditions expected in Australia because of climate change (e.g., Seaton et al. 2015), high-elevation snow-gum dieback may become more prevalent and/or severe in the future, offering little opportunity for trees to recover from dieback events.

Fire

European graziers introduced fires into the ecosystem, and greatly increased fire frequency (e.g., fire-free intervals of 10-20 years) (Good 1982) for pasture management purposes (Zylstra 2006). Cessation of grazing practices (and associated use of fire by graziers) led to the recovery of the ecosystem type. Since this time, wildfires have sporadically affected *Alpine-subalpine Woodlands and Forest*, with some patches burning three or more times over a 10-year period (Fairman et al. 2017). Just 0.5% of Victorian *Alpine-subalpine Woodland and Forest* remains unburnt since 1939 (Morgan et al. 2024).

Modifications to fire severity or frequency are primary threats. Relatively low intensity fires may kill characteristic tree species of *Alpine-subalpine Woodland and Forest* (e.g., *Eucalyptus pauciflora* subsp. *niphophila*) (Good 1982). Increasing fire frequency can change open woodlands with large, old trees to closed, mallee-type communities (Pickering & Barry 2005), increases tree mortality (Fairman et al. 2017), reduces seedling recruitment (Fairman et al. 2017), decreases available hollows in characteristic tree species (e.g., *E. pauciflora*; Salmona et al. 2018), and shifts shrubby understories to those dominated by grasses (Fairman et al. 2017). Structural and compositional changes to *Alpine-subalpine Woodland and Forest* arising from altered fire regimes may have flow on effects to fauna occupying these ecosystems (e.g., mammal richness and site occupancy tend to be highest in long-unburnt woodland and forest; Dixon et al. 2019).

Weeds

Weed species have been documented in alpine-subalpine areas since the late 1800s (Maiden 1898). Subsequent introductions have occurred via historical grazing practices; construction of hydroelectric, ski resort, road and track infrastructure; cultivation of ornamental gardens around ski lodges; slope stabilization and historical rehabilitation practices (Johnston & Pickering 2001; Schroder et al. 2015). The number of weed species in alpine and subalpine zones has increased considerably, since the late 1800s, with 140 species documented across the Australian Alps in the late 1990s (ACT: n = 10, NSW: n = 165, Victoria: n = 117)(Johnston & Pickering 2001). While natural communities in the Australian Alps are ostensibly resistant to weed invasion (rarely greater than 5% cover) (McDougall et al. 2005), *Alpine-subalpine Woodland and Forest* in Northern regions of the Kosciuszko National Park has been observed to contain a greater cover of exotic plant species than lower elevation *Eucalyptus*-dominated forest (Godfree et al. 2004). Under altered climatic regimes (i.e., increased temperatures, reduced snow cover and duration), weeds are predicted to increase in richness, cover and extent in alpine and subalpine zones (Bear et al. 2006).

Impacts associated with invasion by weeds include alterations to soil structure and soil moisture (Waterhouse 1988), reductions in light availability through shading (Waterhouse 1988), and competition with native species (Johnston 2006). These alterations to environmental and biotic processes lead to the decline in native species diversity (Wearne & Morgan 2004; Hosking & Schroder 2013).

Grazing

In the early 1800s, domestic sheep and cattle grazing was introduced to subalpine areas of Australia (Costin et al. 1979). Rabbits, hares, and deer were subsequently introduced in the late 1800s (Costin et al. 1979). Domestic grazing ceased in NSW in 1958 to address concerns about soil erosion and altered hydrology leading to the long-term recovery of vegetation (K. McDougall pers. comm.) and in 2014 domestic livestock were permanently withdrawn from Victorias's high mountain grazing leases. However, horses (*Equus caballus*), red deer (*Cervus elaphus*), rusa deer (*Cervus timorensis*), sambar deer (*Cervus unicolor*), fallow deer (*Dama dama*), cattle (*Bos taurus*), pigs (*Sus scrofa*), goats (*Capra hircus*), European rabbits (*Oryctolagus cuniculus*), brown hares (*Lepus europaeus*) and hog deer (*Axis porcinus*) continue to occur in *Alpine-subalpine Woodland and Forest* (Green & Osborne 2012; Cairns & Robertson 2015; Claridge 2016). Populations of feral herbivores continue to increase, at least on mainland Australia (Green & Pickering 2013; Claridge 2016; Driscoll et al. 2019).

Impacts associated with grazing by these species include: weed proliferation; reduced litter layer; reduced vegetation height, density, and cover; changed species richness; altered competitive dynamics; altered food availability; altered soil biota; increased siltation; variability in water flow; increased soil compaction, erosion, and exposure; altered soil temperature and moisture; and altered ecosystem processes. Further, grazing by feral herbivores will likely reduce resilience to, and recovery from, other disturbances.

Habitat Loss and Fragmentation

Habitat loss and fragmentation within *Alpine-subalpine Woodland and Forest* are primarily caused by the construction of ski resorts, utilities (e.g., electrical transmission lines and hydroelectric power schemes), road infrastructure, and historic grazing practices (e.g., tree clearing). Significant construction activities occurred in the late 1940s with the construction of the Snowy Mountains Hydroelectric Power Scheme (Good 1992), and in the 1960s/1970s with further development of the Snowy Mountains Hydro Scheme (Good 1992), as well as construction of ski resorts (predominantly across the NSW and Victorian Alps) (Worboys & Pickering 2002). Ongoing construction activities (i.e., roads, urban infrastructure, ski infrastructure) occur in high elevation mountain areas to support tourism (where visitation rates and economic contributions to total gross state product are substantial e.g., see Sato et al. 2014; ARCC 2020), and to expand power generation schemes (e.g., Snowy 2.0, an expansion of the Snowy Mountains Hydro Scheme) (EMM Consulting 2018). The construction of ski runs, utilities and roads has involved clearing of large tracts of vegetation (including trees, rocks, logs) (PBPL 2002), with subsequent stabilisation of bare earth using exotic grasses or rehabilitation through native revegetation programs (AALC 2006).

This threat is highly localised, but where it occurs, it is severe and transformative. Across the entire mainland distribution of *Alpine-subalpine Woodland and Forest*, estimates of habitat loss are 1% since 1750 in NSW (Gellie 2005) and 0.1 % in Victoria (DSE 2020a).

Climate Change

Climate change is an overarching threat that influences and alters the dynamics, extent, and severity of other threats. The frequency and severity of fires is likely to increase (Reisinger et al. 2014) with unprecedented events like the 2019/2020 megafires potentially occurring more often (Collins et al. 2022). The duration of drought events may increase, leading to increased stress in trees and their decline if moderate conditions do not return (Dodson 2001). Weeds and non-native herbivores may migrate up mountains and proliferate at higher elevations with increased ambient temperatures (Pickering et al. 2004; McDougall et al. 2005; Schroder 2014); lower elevation plant species may shift to higher elevations, altering the composition of *Alpine-subalpine Woodland and Forest* (e.g., altered eucalypt distributions; (González-Orozco et al. 2016). Tourism-based industries may seek to expand the extent of summer recreation (e.g., mountain bike trails) (Scott & McBoyle 2007) or move winter operations to higher elevations with decreasing snow cover and duration (as has been observed in some European ski resorts) (Scott & McBoyle 2007), leading to greater habitat loss and fragmentation.

IUCN Stresses Classification

1.2 – Ecosystem degradation
1.3 – Indirect ecosystem effects

IUCN Threats Classification

1.3 – Tourism & Recreation Areas (ski areas, tourist resorts, mountain bike trails)
4.1 – Roads & railroads
7.2 – Dams & water management/use (Snowy Hydro)
7.1.1 – Increase in fire frequency / intensity
8.1.2 – Invasive non-native species
8.2.1 – Problematic Native Species / Diseases (*Phoracantha* sp. longicorn beetle)
11.1 – Habitat shifting & Alteration (climate change affecting composition & potentially location)
11.2 – Droughts
11.3 – Temperature Extremes

Ecosystem collapse

Based on the processes/threats driving the system (and whether they act in combination or individually), it is possible that *Alpine-subalpine Woodland and Forest* has four different collapse states (Figure 12):

1) *If non-native ground layer and shrub species become dominant but characteristic native tree species remain*: this would represent a highly degraded woodland form that would likely support a depauperate faunal community. For example, in lower elevation *Alpine-subalpine Woodland and Forest,* grazing history has allowed *Anthoxanthum oderatum* (sweet vernal grass) to outcompete other ground layer species so that it is now dominant (but snow-gum species remain).
2) *If non-native ground layer and shrub species become dominant and trees are removed:* this would represent a degraded, non-woodland collapse state. Examples include cleared ski runs, or areas where anthropogenic infrastructure exists (roads, houses etc.).
3) *If native ground layer and shrubs species remain intact but trees are lost*: this would represent a derived state. An example of this may be where snow-gum dieback kills mature trees but leaves the mid-storey and ground layer plant species intact.
4) *If the ground layer and midstory are predominantly native and trees remain in the ecosystem, but their composition is altered* (i.e., snow-gums are no longer dominant or no longer form part of the community): this would represent an intact but altered woodland or forest ecosystem. For example, with changing climate regimes, it is possible that lower elevation tree species (e.g., Alpine Ash, *Eucalyptus delegatensis*) may advance up mountains into climatically more suitable locations outcompeting species such as *E. pacuiflora* subsp. *niphophila* and *E. pacuiflora* subsp. *debeuzevillei*.

In this assessment, ecosystem collapse of *Alpine-subalpine Woodland and Forest* occurs where:

1) Area: The mapped distribution declines to zero (100% loss) (Criteria A and B),

2) Fire Frequency is greater than 3 in 20 years across the whole ecosystem with no signs of regeneration (Criterion C).

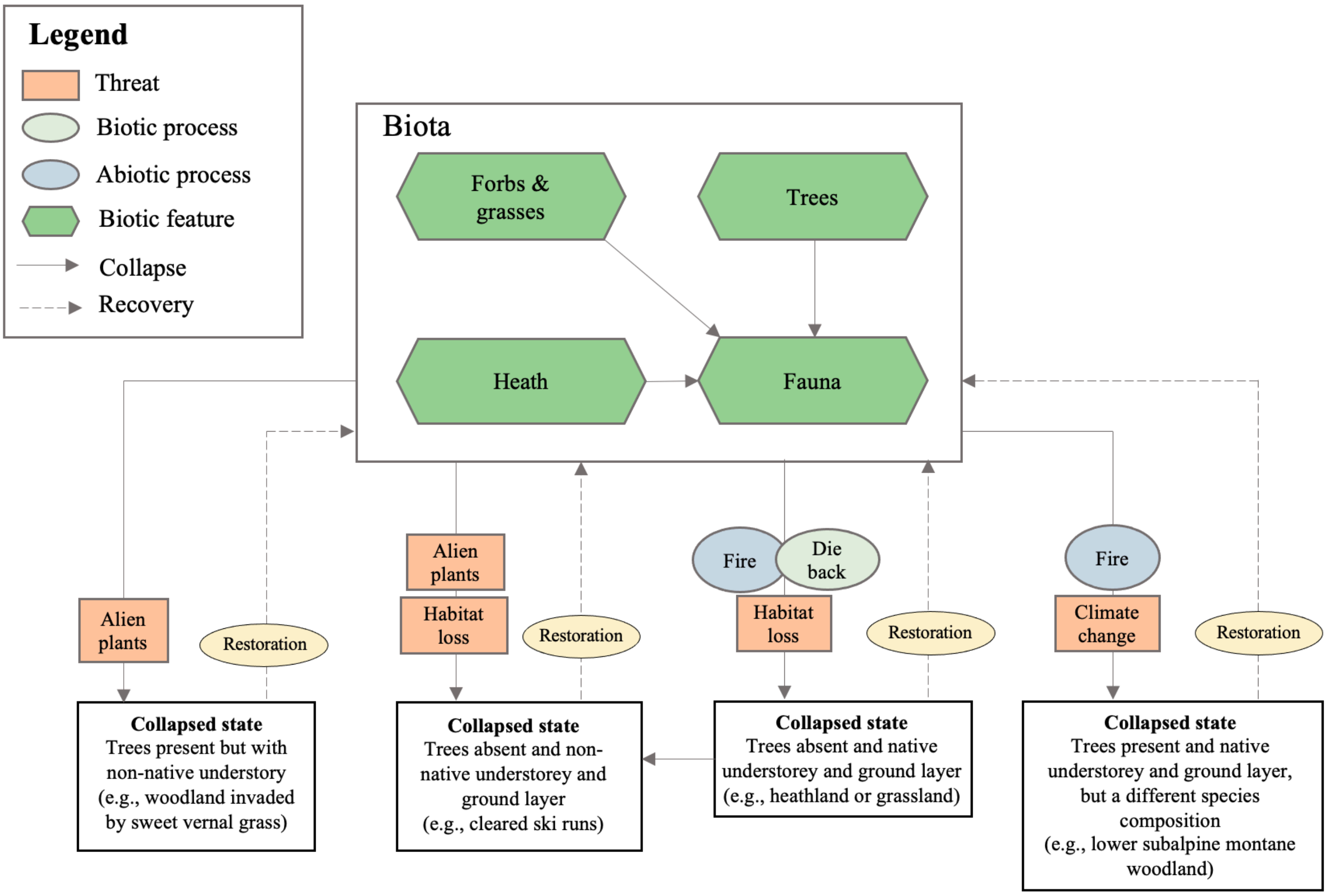


Figure 12. State-and-transition model showing pathways to collapse and recovery (via restoration) for *Alpine-subalpine Woodland and Forest*.

Risk Assessment

Criterion A: reduction in distribution

Summary

Based on available evidence, the ecosystem type is assessed as **Least Concern** under sub-criteria A1, A2, and A3.

Methods

Present-day maps used in analyses were constructed using a combination of existing map products from the Australian Capital Territory (ACT Government 2018), New South Wales (Gellie 2005; DCCEEW 2011a), Victoria (DELWP 2021), and Tasmania (Kitchener & Harris 2013; DPIPWE 2020). Further details regarding construction methods and data used are provided in main methods.

For sub-criterion A1, we assumed that the main causes of ecosystem loss over the last 50 years have been due to ski, road, and energy infrastructure construction. We collated present-day areas of skiable terrain from available literature as an estimate of area lost from ski infrastructure. While spatial data on assets and infrastructure are available, they are point or line data rather than area-based estimates, so we used skiable terrain data only. For sub-criterion A2a/A2b, we assumed that the main causes of loss will be the approximately the same as under sub-criterion A1.

For sub-criterion A3, we determined reductions in ecosystem distribution since ~ 1750 by constructing pre-1750 distributions using existing map products from New South Wales (Gellie 2005; DCCEEW 2011a) and Victoria (DELWP 2021) Pre-1750 map products developed from TASVEG4.0 (DPIPWE 2020) were not available for Tasmania, so these were excluded, and change in Victoria and NSW only were assessed as indicative off general trends.

Assessment outcomes

Recent change (A1)

Based on available literature (Sato et al. 2014; MountainWatch 2023a), the total skiable terrain is 34.84 km^2. Accurate estimates of area for roads and energy infrastructure were not available at the time of assessment. However, even if the skiable terrain estimate was tripled to account for roads and energy infrastructure, the total area lost would (non-conservatively) equate to 104.52 km^2. As the present-day ecosystem is 4,433.18 km^2, the estimated area lost represents (104.52/4433.18) = 2.36% loss in ecosystem area over the past 50 years. As such, the ecosystem type is assessed as **Least Concern** under sub-criterion A1.

Future change (A2)

We assume that the level of impact in the future from infrastructure projects is likely to be similar to that of the past developments outlined in A1. Therefore, the ecosystem type is **Least Concern** under sub-criterion A2.

Historic change (A3)

Based on estimated changes in Victoria and NSW, the distribution of the ecosystem has declined <1% (3511-3513 km2 historical area, for scenarios 1 and 2 respectively, to 3489-3491 km2 current area). Therefore, the ecosystem type is assessed as **Least Concern** under sub-criterion A3.

Criterion B: restricted distribution

Summary

The risk status of *Alpine-subalpine Woodland and Forest* is assessed as **Least Concern** under criteria B1, B2 and B3.

Methods

Under criterion B, we determined the extent of occurrence (EOO; sub-criterion B1) and area of occupancy (AOO; sub-criterion B2) using a combination of existing map products from the Australian Capital Territory (ACT Government 2018), New South Wales (Gellie 2005; DCCEEW 2011a), Victoria (DELWP 2021), and Tasmania (Kitchener & Harris 2013; DPIPWE 2020). The data for subalpine treeless Victoria were augmented by experts with high resolution aerial photography from 2011 to better reflect on-ground ecosystem boundaries. The specific vegetation communities included from each mapping product are listed in the main methods. While there may be some limitations in the on-ground accuracy of these products, they represent the most extensive and/or accurate mapping available within each state for the defined bounds of the *Alpine-subalpine Woodland and Forest* at the time of assessment (May 2020), as advised by representatives from each government department involved in the assessment process.

We calculated EOO using a minimum convex polygon enclosing all mapped occurrences of *Alpine-subalpine Woodland and Forest* in Australia. We calculated AOO based on the number of 10 × 10 km grid cells that contained *Alpine-subalpine Woodland and Forest* (i.e., we did not apply to 1% rule; see Bland et al. 2017). We determined the number of threat-defined locations (sub-criterion B3) by considering historical fire extent (and frequency of fire affecting large proportions of *Alpine-subalpine Woodland and Forest*), as well as connectivity of *Alpine-subalpine Woodland and Forest* patches. We used a combination of present-day *Alpine-subalpine Woodland and Forest* mapping (constructed for criterion B1 and B2), fire history maps for NSW, Victoria and Tasmania, and national-scale woody vegetation mapping (ABARES 2018) to visualise threat-defined locations.

Assessment Outcome

Sub-criterion B1

The Extent of Occurrence (EOO) of *Alpine-subalpine Woodland and Forest* is currently estimated at 164,134.76 km^2 (Figure 13). As this is well above the threshold for Vulnerable, the risk status is **Least Concern** under sub-criterion B1.

Sub-criterion B2

Alpine-subalpine Woodland and Forest occupies 334 10 × 10 km^2 grid cells (Area of Occupancy, AOO) (Figure 13). The risk status is therefore **Least Concern** under sub-criterion B2.

Sub-criterion B3

Fire is the most serious plausible threat to *Alpine-subalpine Woodland and Forest.* The ecosystem type is estimated to occupy between 2 and > 10 threat-defined locations. Two threat-defined locations are possible assuming that: (1) the entire extent of the ecosystem type across ACT, NSW and Victoria is connected, (2) that the entire extent across Tasmania is connected, and (3) that the Australian mainland and Tasmanian woodland extents independently can each be destroyed by a single, high-intensity mega-fire, or multiple high-intensity fires arising from independent ignition points in a single fire season. This is plausible as patches on the mainland are well connected by wooded vegetation (ABARES 2018), and thus there is potential for fire to carry across separate patches of *Alpine-subalpine Woodland and Forest*. In Tasmania, most patches are connected, with a small portion of the ecosystem type isolated in north-eastern Tasmania (i.e., wooded vegetation is sparse between these patches and the portion of the ecosystem type on the Central Plateau of Tasmania). Therefore, it is likely that at least two independent ignition points (and fires) would be required to burn the entire extent in Tasmania. Alternatively, 10 or more threat-defined locations are possible assuming that discrete patches of *Alpine-subalpine Woodland and Forest* (see Figure 13) are independent of one another, thus fire acts independently on each patch (e.g., independent fire ignitions required in each patch to cause collapse in that location).

Fires have affected extensive areas of *Alpine-subalpine Woodland and Forest* multiple times on mainland Australia (e.g., 1938-1939 fire season, 2002-2003 fire season, 2019-2020 fire season) and Tasmania (e.g., 1930s, 1960s, 2012-2013 fire season, 2018-2019 fire season). However, even the most extensive fires did not affect the full extent across the mainland and Tasmania. Intervals of < 20 years between extensive fires has been observed more recently on the mainland (i.e., 2002-2003 and 2019-2020) and in Tasmania (i.e., 2012-2013 and 2018-2019), indicating that there is potential for fires to cause ecosystem collapse or greatly elevate threat status (e.g., to Critically Endangered) within a very short period (i.e., 20 years). However, the probability of multiple, high-intensity fires occurring across the entire extent over the next 20 years is extremely small (see criterion C2). Therefore, the ecosystem type is assessed as **Least Concern** under sub-criterion B3

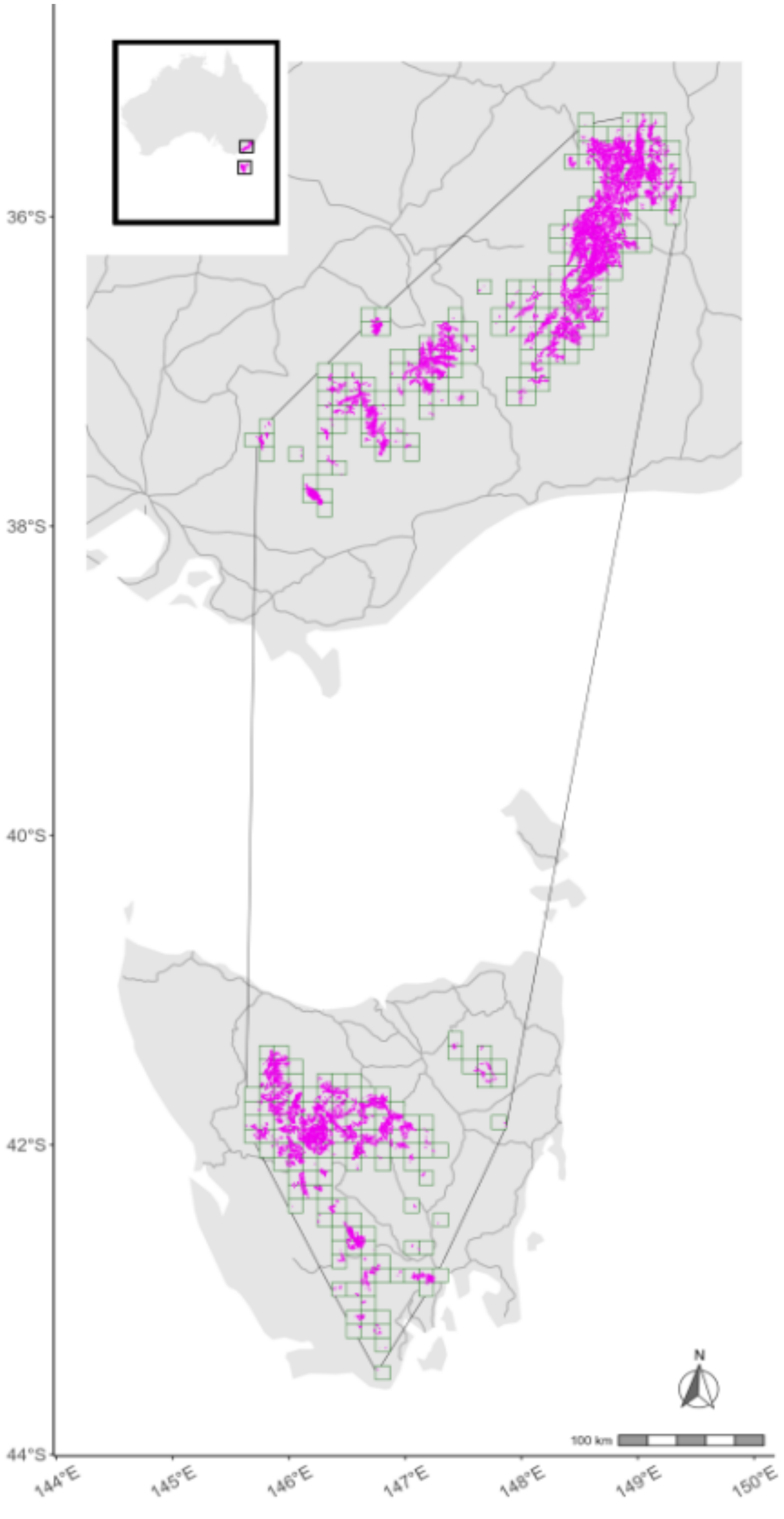


Figure 13. Map of *Alpine-subalpine Woodland and Forest* (magenta polygons) across Australia, showing Extent of Occurrence (black polygon) and Area of Occupancy, where the 1% rule was not applied (green squares).

Criterion C: Environmental degradation

Summary

The risk status for *Alpine-subalpine Woodland and Forest* is **Near Threatened** based on the recent increase in the frequency of fires (sub-criterion C1). However, there was insufficient data to assess past or predict future change in soil moisture, so the ecosystem type is listed as Data Deficient under this indicator.

Identification of abiotic indicators

We examined the relevance and data availability/quality for two indicators to assess the risk of collapse from disruption of abiotic processes and/or interactions:

- *Fire frequency:* a measure of the frequency in which the ecosystem burns within a 20-year period
- *Soil moisture*: indicates potential tree stress

Indicator: Fire frequency

Relevance to ecosystem function

Fire has historically been an infrequent occurrence in *Alpine-subalpine Woodland and Forest*. Snow-gums are sensitive to fire as high-severity fires can kill mature trees, reduce seedling recruitment, shift shrubby understories to those dominated by grasses (Fairman et al. 2017), and decrease available hollows for fauna (Salmona et al. 2018). Fires may also increase invasion of weeds (Scherrer et al. 2004 but see Verrall & Pickering 2019) and the occurrence of insect-related dieback (M. Brookhouse, pers. comm.)

Data availability and quality

Recent (C1) and Historical (C3)

Historical records of bushfires and planned burns are available as shapefiles, detailed in the main methods.

Future (C2a)

We obtained future fire simulations for mainland Australian alpine and subalpine ecosystems for 2060 to 2079 from the landscape fire modelling framework ‘Fire Regime and Operations Tool’ (FROST), detailed in the main methods.

Selection of collapse threshold

While a flush of regeneration may occur post-fire in *Alpine-subalpine Woodland and Forest*, short fire return intervals will lead to death of regeneration and new recruits. Moreover, single high severity fires can lead to up to 15% tree mortality (Coates et al. 2012). The < 20-year interval between extensive fires observed recently on the mainland (i.e., 2002-2003 and 2019-2020) and in Tasmania (i.e., 2012-2013 and 2018-2019) indicates that there is potential for fires to cause ecosystem collapse or greatly elevate threat status of *Alpine-subalpine Woodland and Forest* (e.g., to Critically Endangered) within a very short period (i.e., 20 years). Therefore, fire frequencies of ≥

2 fires in 20 years (equating to a spatially weighted annual probability of burning of 0.10) with no signs of regeneration of characteristic vegetation would indicate ecosystem collapse.

Selection of initial and present/future values

Recent change (C1)

To define the initial and present fire frequency, we analysed the fire history datasets from the NSW and the ACT, Tasmanian, and Victorian state governments (DSE 2020b; NSW Goverment 2022; Forest Fire Management Victoria 2023; Tasmanian Government 2024). We overlayed the fire history layers and extracted areas where the ecosystem type had been burnt. To generate a time-series, we calculated the number of times and area burnt for 1960-1980 (initial value), 1980-2000, and 1970-2020 (current value). We used these values to calculate the mean annual probability of burning based on the spatially weighted mean area of the ecosystem type burnt at each frequency in each timeframe.

Future change (C2)

We used FROST to create 100 replicate simulations of the 20-year period 2060 to 2079. For each simulation, we extracted a spatial raster that displayed the number of fires burnt within the 20-year period, for each 180 m^2 cell. For each climate scenario, we calculated the percentage of the ecosystem type (based on the number of cells) that burnt at a frequency that exceeded the collapse threshold (i.e., where relative severity is 100%).

Historic change (C3)

We used the same approach as used in sub-criterion C1 to calculate initial and present values. In lieu of historical data, we assumed that the fire frequency in 1939-1960 was relatively representative of the fire frequency c. 1750.

Calculation of severity and extent

Recent change (C1)

Between 1970 and 2020, 49 fires have burnt approximately 3,269 km^2 (74%) of the ecosystem type. Overall, 71.7% of the ecosystem type has burnt once within the past 20 years (Table 7). None of the ecosystem type has burnt at frequencies resulting in ecosystem collapse (≥ 2 fires in 20 years). However, the area burnt greatly increased from 1960-1980 (19% burnt) to 2000-2020 (71.7%). This equates to an increase from 0.009 to 0.035 in the spatially weighted annual probability of burning. The collapse threshold of ≥ 2 fires in 20 years has a spatially weighted annual probability of burning of 0.10. Thus, the relative severity equates to 28.57% [((0.009-0.035) / (0.009-0.1)) × 100]. Thus, this ecosystem type is assigned a risk status of **Near Threatened** under criterion C1.

Table 7. The percentage (%) of the *Alpine-subalpine Woodland and Forest* area (based on the number of cells) that have burnt over the threshold from 1960-1980, 1980-2000, and 2000-2020, and the actual respective fire frequencies.

Period of 10-year interval	**Percentage of ecosystem burnt over threshold (2/10 years)**	**Number of fires**	**Percentage of ecosystem burnt**	**Annual probability of burning (spatially weighted mean)**
1960-1980	0%	1	19.04%	0.9
1980-2000	0%	1	19.69%	1
2000-2020	0%	1	71.73%	3.5

Future change (C2)

Under all four climate scenarios, there were low predicted frequency of wildfires and most of the extent was not predicted to burn at a frequency that exceeded the collapse threshold (Table 8). The mean predicted fire frequency was less than the collapse threshold for the entire mainland extent (Figure 14). Overall, the probability of any part of the ecosystem type (i.e., at least one grid cell) burning at a frequency that met or exceeded the collapse threshold ranged from 6% (climate scenario CSIRO-R1) to 9% (CSIRO_R3). However, the extent of the ecosystem type that exceeded the collapse threshold (i.e., relative severity = 100%) in one or more simulations was small, ranging from 0.9% (CSIRO-R1) to 6.8% (ECHAM-R2). Very small parts of the mainland extent displayed a very low relative severity of progress towards the collapse threshold. This is likely to also be representative of the Tasmanian parts of the ecosystem type, resulting in a risk status of **Least Concern** under sub-criterion C2.

Table 8. The percentage (%) of the ecosystem type area (based on the number of grid cells) and the number of simulations (out of 100) that burnt at a frequency that met or exceeded the collapse threshold for each climate scenario.

Number of simulations	**CSIRO-R1**	**CSIRO-R3**	**ECHAM-R1**	**ECHAM-R2**
0	99.1%	95.5%	94.27%	93.29%
1	0.01%	0.14%	0.05%	0.04%
2	0.75%	3.76%	4.85%	5.78%
3	-	0.02%	<0.01%	<0.01%
4	0.07	0.50%	0.67%	0.78%
5	-	0.01%		-
6	<0.01%	0.07%	0.14%	0.10%
7	-	-<0.01%	-	0
8	-	-	0.02%	0.01%-
9	-	<0.01%	-	

Historical change (C3)

According to fire records, very little of the ecosystem type was burnt between 1939 and 1960 (0.8%). This would equate to a spatially weighted annual probability of burning of 0.016. This compares to current records, which suggest 71.73% of the ecosystem type has burnt at least once since 2000, equating to a relative severity of 35.47% [((0.016-14.2)/(0.016-40)) × 100]. However, Australian fire records prior to 1960 are unreliable, and this large transformation in area burnt cannot be reliably attributed to changes in the fire regimes. Therefore, the risk status under Criterion C3 is assessed as **Data Deficient.**

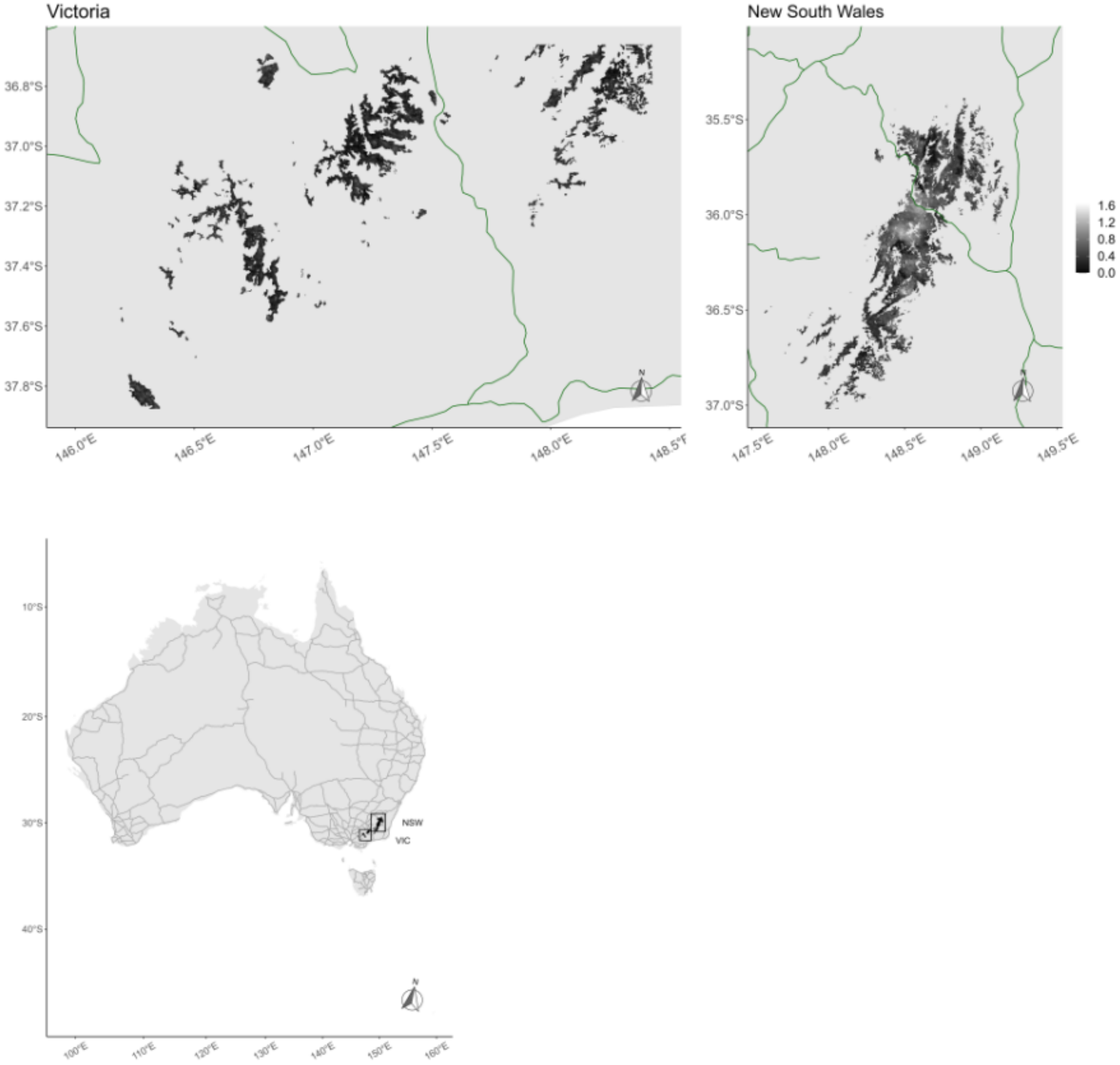


Figure 14. Mean projected future fire frequency (2060-2079). Mean is calculated across 100 replicated simulations.

Indicator: Soil moisture

Successive summer periods with low soil moisture may stress snow-gums, resulting in direct mortality of trees (Nolan et al. 2021) or make them susceptible to insect attack due to hydraulic stress (Hartmann et al. 2018). Soil moisture data has been modelled for past, current and future scenarios across the entire extent of the ecosystem type (see AWAP and NicheMapR) (Jones et al. 2009; Kearney & Porter 2017). However, there is currently insufficient information to develop soil moisture thresholds that trigger individual tree death or insect attack for key eucalypt species, or that might lead to collapse of the ecosystem type. As such, the ecosystem is **Data Deficient** under this indicator.

Criterion D: Disruption of biotic processes and/or interactions

Summary

This ecosystem type is relatively data-poor, and data are unavailable to reliably assess this criterion. The risk status is **Data Deficient** under all sub-criteria.

Identification of biotic indicators

We examined the relevance and data availability/quality for three indicators to assess the risk of collapse from disruption of biotic processes and/or interactions:

- *Dead tree density or abundance*: as a measure of dieback
- *Weed cover*: the percentage of overlapping cover of weeds
- *Exotic herbivore density*: the density or abundance of exotic herbivores

Indicator: Dead tree density or abundance

Relevance to ecosystem function

Eucalyptus trees such as *Eucalyptus pauciflora* subsp. *niphophila, E. coccifera*, *E. gunnii* are the defining features of *Alpine-subalpine Woodland and Forest* (Kitchener & Harris 2013; Venn et al. 2017). Dieback of these tree species is a natural process that occurs when drought conditions persist for extended periods, causing the water stressed trees to be more susceptible to insect attack and death (Seaton et al. 2015). *Eucalyptus* dieback is characterised by a decline in the health of the canopy (browning and thinning of leaves), followed by canopy death. Subsequently, individual stems die, followed by entire trees and – in severe cases – entire stands (SOSnowgum 2023). The stages of canopy and tree death are couple with signs of insect activity including clean cut circular holes with frass (chewed wood) and/or kino (deep red exudate), puckering or dying bark, and horizontal feeding galleries (SOSnowgum 2023).

Data availability and quality

Aerial photography in 1964, 1978 and 1988 revealed expanding historical dieback events in NSW (Shields 1993). Recent observations indicate that snow-gum dieback is currently widespread throughout the Australian Alps (SOSnowgum 2023), particularly between 1600 m to 1800 m above sea level. The increase in dieback is partly attributed to an outbreak of wood-boring longicorn beetles (Brookhouse, pers. comm) that ringbark trees, impairing flow of water and carbohydrates around the tree, ultimately leading to tree death. A study conducted in New South Wales ski resorts indicated that dieback appears to occur randomly (except in locations like Thredbo and Charlotte Pass ski resorts where higher elevations were more affected) (DPIE 2021). The predicted increase in frequency, intensity, and duration of drought conditions under climate change (e.g., Seaton et al. 2015) may increase the frequency of insect infestation and subsequently dieback, putting the ecosystem type at risk of collapse.

While research is active on this subject, there are currently insufficient data (beyond sightings collated through SOSnowgum and the High Country Dieback Network, and the DPIE ski resort mapping project; see (DPIE 2021; SOSnowgum 2023) to analyse risk of collapse due to dieback, and no projections of future dieback events (or recovery potential from current dieback events). The DPIE ski resort mapping project used remote sensing techniques and high-resolution aerial imagery to quantify dieback extent and severity in NSW ski resorts. The study indicated that in 2021, 613 ha of *Alpine-subalpine Woodland and Forest* in ski resorts (i.e., 97% of the ecosystem type within resorts) was affected by dieback, with 154.8 ha (or 24.5% of the ecosystem type) suffering from severe dieback. However, seedling and crown regrowth in dieback areas was identified and true extent of recovery requires further investigation. Based on these data, dieback is a considerable threat that can affect a large extent of *Alpine-subalpine Woodland and Forest* and warrants further investigation. However, the ecosystem type is currently **Data Deficient** under this indicator.

Indicator: Weed cover

Relevance to ecosystem function

Weed invasions are a threat to *Alpine-subalpine Woodland and Forest*. Weeds can alter the soil structure and soil moisture (Waterhouse 1988), reduce light availability (Waterhouse 1988), and out-compete native species (Johnston 2006), leading to the decline in native species diversity (Wearne & Morgan 2004; Hosking & Schroder 2013).

Data availability and quality

Weed species have increased in the alpine and subalpine zone since the late 1800s (Johnston & Pickering 2001). While intact ecosystems tend to be relatively resilient to invasion (McDougall et al. 2005), *Alpine-subalpine Woodland and Forest* has greater cover of exotic plant species than lower elevation *Eucalyptus* forest (Godfree et al. 2004; in Northern Kosciuszko National Park). The warmer temperatures and reduced snow cover and duration predicted under climate change are expected to enhance the richness and extent of weeds in alpine and subalpine zones (Bear et al. 2006). At lower subalpine elevations, there are instances of invasion and extensive establishment of weed species in the ground storey of *Alpine-subalpine Woodland and Forest* by *Anthoxanthum odoratum* (sweet vernal grass; K. McDougall, pers. comm.). Future climate modelling also highlighted that alpine-subalpine areas will increasingly support conditions that allow for weed establishment and expansion (Duursma et al. 2013; WeedFutures 2023).

Data on weed cover (particularly of significant, transformational weeds) are not available across the full extent of this ecosystem type. Further, the ability for weeds to invade (and establish) requires further investigation (Verrall & Pickering 2019), as well as thresholds of weed cover that cause severe ecological dysfunction and collapse. As such, the ecosystem type is **Data Deficient** under this indicator.

Indicator: Exotic herbivore density

Relevance to ecosystem function

Exotic herbivores such as horse, sheep, cattle, deer, pigs, goats, rabbits, and hares can substantially degrade *Alpine-subalpine Woodland and Forest*. Grazing and trampling by exotic herbivores can increase soil erosion (Duretto 2018; Ward-Jones et al. 2019), alter the characteristic hydrology and chemical properties of the soil (Duretto 2018), enhance weed invasions, and alter the native vegetation cover, structure, and composition (Duretto 2018; Ward-Jones et al. 2019).

Data availability and quality

Domestic grazing ceased in 1958 in NSW (leading to a recovery of vegetation; K. McDougall pers. comm.) and in 2014 in Victoria (Costin et al. 1979), yet feral herbivores continue to increase in abundance on mainland Australia (Green & Pickering 2013; Claridge 2016; Driscoll et al. 2019). In NSW, mapping indicates that deer, horse, and pig distributions have likely expanded across the alpine-subalpine region since 2009 (DPI 2023), and that there are areas where deer, horse, and pig abundance is (or has been) considered “high” (DPI 2023). However, there is currently insufficient information on the density of exotic herbivores, the density required to cause collapse, and their impact across the full extent of the ecosystem type, to assess risk of collapse. As such, the ecosystem type is **Data Deficient** under this indicator.

Criterion E: Quantitative analysis of probability of collapse

Assessment outcome

No stochastic models of *Alpine-subalpine Woodland and Forest* are currently available and there are presently insufficient data to reliably inform simulations key collapse indicators (dieback, interactions between fire and dieback, effects of soil moisture, snow duration and drought on stand structure). Therefore, the risk status is **Data Deficient** under criterion E.

Alpine-subalpine Open Grassy Heathland and Herbfield

Authors

Clare Vernon, Chloe Sato, Jessica Rowland, Emily Nicholson, Ella Plumanns Pouton, Kate Umbers

Reviewers

James Camac, Brodie Verrall

Contributors

James Camac, John Morgan, Richard Williams, Keith McDougall

Biome

T6 Polar-alpine

Functional group

T6.4 Temperate alpine grasslands and shrublands

IUCN status

Least Concern (Least Concern – Near Threatened)

Assessment Summary

Alpine-subalpine Open Grassy Heathland and Herbfield is dominated by grasses, forbs, herbs, and sedges, with sparse shrub coverage (typically 0–30%). It is restricted to elevations > 1100 m on the mainland and > 600 m in Tasmania. The status of *Alpine-subalpine Open Grassy Heathland and Herbfield* is **Least Concern (Least Concern – Near Threatened)** (Table 9). Although predicted warmer, drier climate conditions are likely to promote shrub thickening and expansion of adjacent *Alpine-subalpine Closed Heath* into *Alpine-subalpine Open Grassy Heathland and Herbfield*, this is unlikely to threaten ecosystem persistence in the near future.

Table 9. Summary of the IUCN Red List of Ecosystems assessment of *Alpine-subalpine Open Grassy Heathland and Herbfield.*

Criteria	**A**	**B**	**C**	**D**	**E**	**Overall**
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	LC	LC	LC (LC-NT)	DD	LC (LC-NT)
Sub-criterion 2 *A, C, D: 50-year period including present & future* *B: AOO*	LC	LC	LC	LC		
Sub-criterion 3 *since ~1750* *B: number of locations*	LC	LC	DD	LC		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to Criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-Criteria

Ecosystem Description

Classification

In the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022a), *Alpine-subalpine Open Grassy Heathland and Herbfield* is a Level 6 sub-global ecosystem type, belonging to Ecosystem Functional Group *T6.4 Temperate alpine grasslands and shrublands* within the *Polar/alpine (Cryogenic)* biome. Under the National Vegetation Information System (NVIS) 6.0, the ecosystem type aligns with *Major Vegetation Group (MVG) 19 (Tussock grasslands)*, *MVG 30 (Heathlands)*, *Major Vegetation Subgroup (MVS) 18 (Heathlands)*, *MVS 30 (Montane and alpine heathlands)* and *MVS 36 (Alpine grasslands and herbfields*) (Keith & Pellow 2015; Australian Government Department of Agriculture Water and Environment 2020).

Alpine-subalpine Open Grassy Heathland and Herbfield encompasses several vegetation communities across the mainland that represent various communities dominated by grass, forbs, and occasional shrubs (McDougall & Walsh 2007; Appendix 1). In New South Wales (NSW), *Alpine-subalpine Open Grassy Heathland and Herbfield* is represented by Vegetation Groups *129 (Alpine tall herbfield)*, *131 (Sub-alpine grassland)*, *132 (Sub-alpine dry herb grassland)*, *206 (Short alpine herbfield)* and may meet the classification of *207 (Alpine rocky low open heathland)* (Mackey et al. 2015). In Victoria, the ecosystem type is classed and mapped as *Ecological Vegetation Classes (EVC) 206 (Sub-alpine grassland)*, *905 (Alpine short herbland)*, *1001 (Alpine grassland)*, *1004 (Alpine grassy heathland)* and *1005 (Alpine grassy heathland/Alpine grassland mosaic)* (Mackey et al. 2015), and is referred to as "tussock grasslands", "tall alpine herbfield" and "heath" in Venn et al. (2017). In Tasmania, the ecosystem type aligns with tussock grasslands, tall alpine herbfield and heath (Venn et al. 2017), and vegetation communities *GPH (Highland Poa grassland)* and *HUE (Eastern alpine vegetation undifferentiated)* (Harris & Kitchener 2005).

Distinction from similar ecosystem types

Alpine-subalpine Open Grassy Heathland and Herbfield is characterised by a dominant understorey of grasses from the *Poaceae* family, forbs, and herbs (Williams & Ashton 1988) (Figure 15) and a short, sparse shrub canopy (0.2–0.5 m; 0–30% cover) made up predominantly of species that do not resprout post-fire (obligate seeders). *Alpine-subalpine Open Grassy Heathland and Herbfield* typically occurs on shallow slopes and at higher elevations where colder, harsher conditions constrain the growth and establishment of woody vegetation tissues.

In contrast, *Alpine-subalpine Closed Heath* occurs on steep and exposed landscapes of mainland Australia, usually at lower elevations, and is characterised by a denser, taller shrub canopy (0.5 – 2.0 m, ≥ 70 % coverage) of both post-fire resprouting and obligate seeder species, with a limited understory of grasses, forbs, and herbs (Williams & Ashton 1988).

Snowpatch Herbfield is characterised by late-lying snow during the warmer months, supressing the establishment of woody flora and thus usually dominated by non-vascular plants and herbs (Wahren et al. 2001; Williams et al. 2015). *Tasmanian Alpine Heath* is limited to Tasmania and is dominated by endemic shrubs that form both open and closed canopies with a dense understorey of rushes, sedges and forbs, and a notable absence of fire disturbances (Kirkpatrick & Bridle 2013; Venn et al. 2017).

Alpine-subalpine Woodland and Forest is found adjacent to *Alpine-subalpine Open Grassy Heathland and Herbfield* but at lower elevational bands and is characterised by a canopy of small

(< 12 m) *Eucalyptus pauciflora* with an understorey of grasses, herbs and shrubs (Figure 15) (Harris & Kitchener 2005; Venn et al. 2017).

Figure 15. Left: Poa valley grassland in the Happy Jacks area, Kosciuszko National Park. Photo: Keith McDougall. Middle: Open grassy heathland at Mt Nelse West Peak, Victoria. Photo: Clare Vernon. Right: herbfield in foreground mosaics with open heathland, closed heathland (midground) and grassland (background) at Mt Nelse North Peak, Victoria. Photo: Clare Vernon.

Distribution

Alpine-subalpine Open Grassy Heathland and Herbfield occurs in the mountainous regions of south-eastern Australia, in the alpine and the treeless subalpine zones on a wide range of landforms, from plateaus to cliffs, ridges and block fields (Harris & Kitchener 2005; Venn et al. 2017). Derived extents also occur in Tasmania: these regions were historically *Alpine-subalpine Woodland and Forest* and *Tasmanian Alpine Heath* where the dominant shrub and tree overstorey was cleared using fire (Harris & Kitchener 2005). *Alpine-subalpine Open Grassy Heathland and Herbfield* occurs between -146.43° and -148.88° longitude and -35.58° and -37.45° latitude. It occupies an altitudinal band of 1000 m to 2200 m above sea level on the mainland (McDougall & Walsh 2007) and > 600 m in Tasmania (Harris & Kitchener 2005). *Alpine-subalpine Open Grassy Heathland and Herbfield* occupies 620.701 km^2, including 465.44 km^2 in NSW, 0.93 km^2 in the ACT, 96.16 km^2 in Victoria and 68.66 km^2 in Tasmania (Figure 16). Spatial products used to develop distribution maps of *Alpine-subalpine Open Grassy Heathland and Herbfield* used the most extensive and/or accurate mapping available at the time of assessment (May 2020), as advised by government representatives involved in the assessment process.

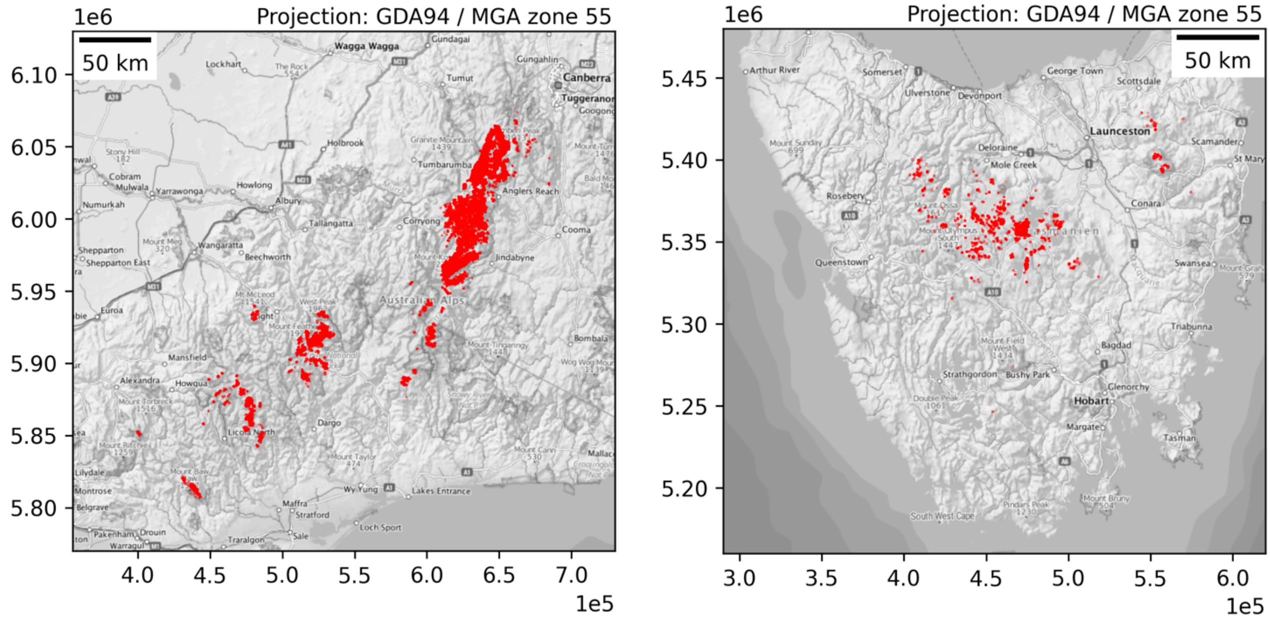


Figure 16. Distribution of *Alpine-subalpine Open Grassy Heathland and Herbfield* (red) on mainland Australia (left) and across Tasmania (right).

Abiotic environment

Alpine-subalpine Open Grassy Heathland and Herbfield is characterised by low annual temperatures (< 8°C, range: 3 - 12°C) (Venn et al. 2017) and high annual precipitation (average annual rainfall 606-2344 mm). Across the mainland extent, snowfall typically occurs during winter and spring (June to September) (Venn et al. 2017), and there is no distinct dry season at higher elevations (Stern et al. 2000). In contrast, the Tasmanian extents remain relatively snow-free throughout the year, with limited snowfall during winter (Stern et al. 2000; Venn et al. 2017).The spring snow melt provides an important source of groundwater recharge for *Alpine-subalpine Open Grassy Heathland and Herbfield* (McDougall et al. 2015). Winter snowfall also acts as an insulating groundcover and canopy layer, preventing freezing of the soil mineral layer, vegetation foliage and roots, and soil fauna (Green & Osborne 1994). Severe winds and frosts are common (Williams et al. 2014; Venn et al. 2017).

On the mainland, soils vary from deep and well drained (up to 1 m) (Williams et al. 2014; Venn et al. 2017) to exposed, rocky, and shallow (Mackey et al. 2015). In Tasmania, soils are typically shallow with basalt and limestone dolerite substrates (Harris & Kitchener 2005). Bare ground cover is generally low (< 3%, range: 3 - 5%) (Camac et al. 2013; Williams et al. 2014) but is promoted by fire, drought, and herbivory.

Characteristic native biota

Alpine and subalpine ecosystems exist in a mosaic with other ecosystem types, and thus species and characteristic floristic communities may occur in several ecosystems. *Alpine-subalpine Open Grassy Heathland and Herbfield* is characterised by grasses, forbs, herbs and sedges, with limited shrub coverage (≤ 30% typical although up to 50% is possible) (Williams et al. 2014) and few trees (Williams et al. 2006; Camac et al. 2013) (Figure 17). The ecosystem frequently intergrades with

Alpine-subalpine Woodland and Forest, *Alpine-subalpine Closed Heath* along elevational and climatic gradients (Figure 17).

Figure 17. Left: *Poa* grassland intergrades with shrubs and trees near Mt Hotham, Victoria. Photo: Clare Vernon, 2024. Right: *Poa* grassland in a cold air drainage line with *Eucalyptus pauciflora* on higher elevation slopes near Tumut, New South Wales. Photo: Clare Vernon, 2024.

Alpine-subalpine Open Grassy Heathland and Herbfield is characterised by grasses, forbs, herbs and sedges, with limited shrub coverage (≤ 30 %, although up to 50 % may be possible, see Williams et al. 2014) and few trees (Williams et al. 2006; Camac et al. 2013) (Figure 15, 18). Species composition varies throughout the ecosystem type (Armstrong et al. 2013). Mainland extents are dominated by tussock-forming snow grasses from the *Poacaeae* family (e.g., *Poa costiniana, P. fawcettiae, P. hiemata, P. labillardieri* and *P. sieberiana*), kangaroo grass (*Themeda triandra*), sedges (e.g., *Luzula* spp., *Carex* spp.), herbs (e.g., *Celmisia* spp., *Craspedia* spp., *Podolepis robusta*) and small dwarf and prostrate shrubs 0.2 - 0.5 m tall (predominantly *Grevillia* spp. and *Phebalium spp*, although other shrubs may occur [e.g., *Asterolasia trymalioides, Baeckea gunniana* and *Hovea montana*]) (Williams et al. 2006; Camac et al. 2013; Mackey et al. 2015; Venn et al. 2017) (Figure 18). In Tasmania, *P. gunnii* and *P. labillardierei* dominate, with forbs, herbs and shrubs present in inter-tussock spaces. The occasional *Eucalyptus spp.* is also present (Harris & Kitchener 2005) (Figure 18).

Figure 18. Left: *Alpine-subalpine Open Grassy Heathland and Herbfield* in Victoria. Photo: James Camac, 2021. Right: Highland *Poa* grassland at White Marsh, north-western Tasmania. Photo: Louise Gilfedder in Harris and Kitchener, 2005.

During the warmer months, some fauna migrate into this ecosystem type for food. These include native migratory insect herbivores and pollinators (*Agrotis infusa* (bogong moth)), many of which are endemic or threatened (e.g., *Kosciuscola spp.* [grasshoppers], Plecoptera [stoneflies] and numerous Lepidoptera including *Hesperiidae spp.* [skippers], *Papilionidae* [swallowtails] and *Psychidae* [case moths]). Migratory insects provide a vital seasonal food source for predatory insectivores (e.g., *Anthus novasellandiae* [Australasian pipit]) and raptors (e.g., *Falco cenchiroides* [Nankeen kestrel]) (Green & Osborne 2012). Characteristic fauna also includes reptiles (e.g., *Liopholis guthega*, Guthega skink) and mammals (including several species of Muridae, such as *Mastacomys fuscus* [broad-toothed rat]). Larger macropods usually migrate during spring and summer on the mainland but are common during winter too in Tasmanian extents (e.g., *Macropus rufogriserus* [red-necked or Bennett's wallaby]) (Green & Osborne 2012).

Key components, processes, and interactions

Cold, snowy climate

Alpine-subalpine Open Grassy Heathland and Herbfield is characterised by a harsh winter climate. Most fauna either migrate to lower, warmer altitudes or remain *in-situ* under the winter snowpack (Green & Osborne 1994; Figure 19). Thick snow insulates vegetation and soil from cold temperatures, frost (and frost heave), wind erosion and desiccation. The accumulation of snowpack also moderates soil moisture, microbial processes and nutrient cycling in soils, and atmosphere energy exchanges by increasing the surface albedo (Green & Osborne 2012; Wilson et al. 2021) . The winter snowpack also moderates the growth of woody flora, creating cold temperatures that limit the thermal energy necessary to support woody growth, and reducing the duration of sunlight available for photosynthesis (Lim et al. 2017; Auld et al. 2022).

Warming temperatures during spring and summer determine the onset of snowmelt, providing an important release of nutrients and water to support the growing season of most flora and fauna (Green & Osborne 2012; Venn et al. 2017). These seasonal windows are an important food source for native migratory species, and higher trophic level species (e.g., raptors) migrate in during these warmer seasons to prey on migratory herbivores and insectivores (e.g., *Lomera caespitosae* [case moth] is the preferred food source of *Falco cenchiroides* [Nankeen kestrel]) (Green & Osborne 2012). While uncommon on the mainland distribution, larger macropods such as *Macropus rufogriserus* (red-necked or Bennett's wallaby) are found in Tasmanian extents: sustained herbivory maintains a simplistic *Poa spp.* vegetation community with a uniform, short structure (Bridle & Kirkpatrick 1999).

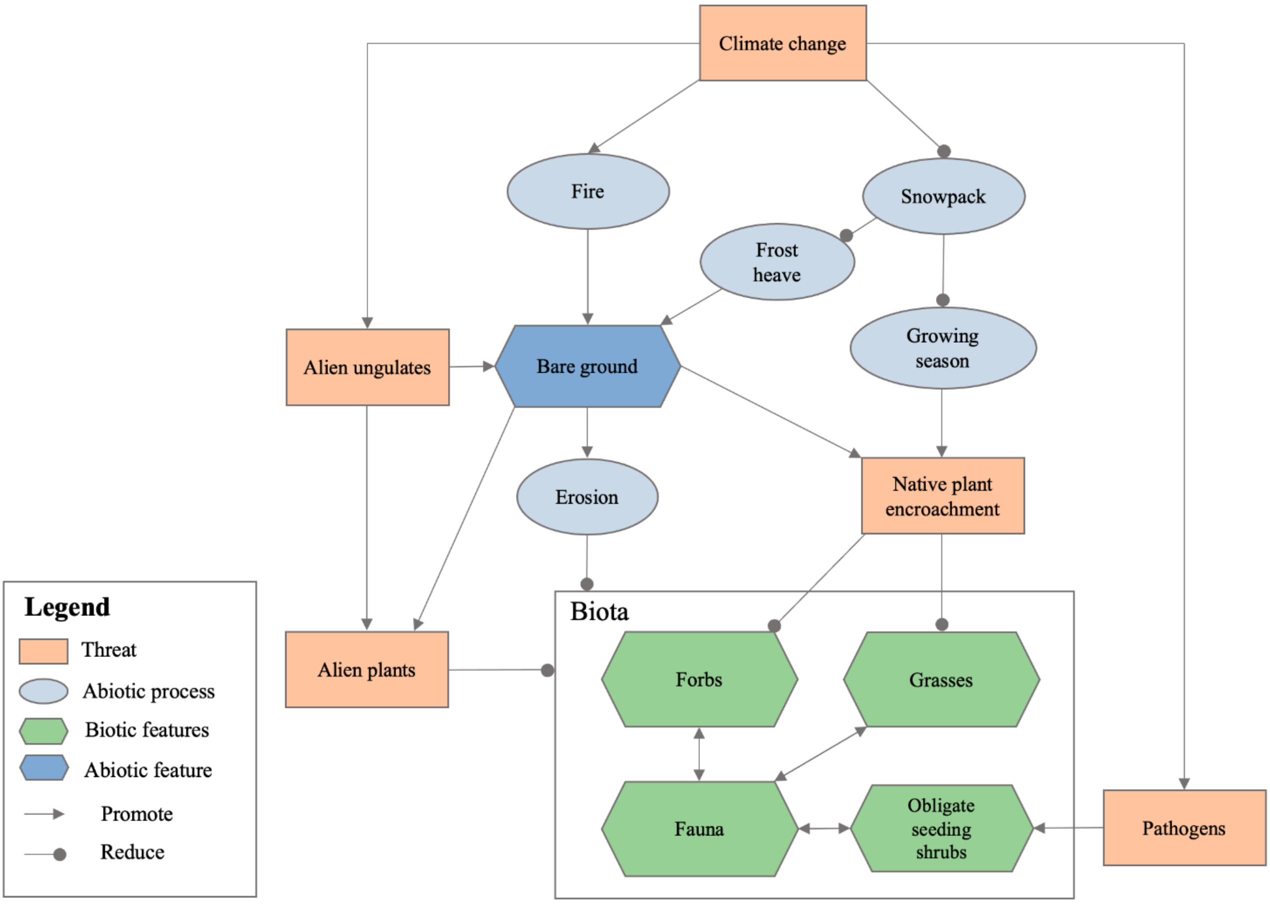


Figure 19. Cause-effect conceptual model for *Alpine-subalpine Open Grassy Heathland and Herbfield.*

Fire

Fire is an important process in the life cycle of vegetation within mainland extents of *Alpine-subalpine Open Grassy Heathland and Herbfield* (McDougall & Walsh 2007; Camac et al. 2013) (Figure 19). Fire is a characteristic, low-frequency disturbance in *Alpine-subalpine Open Grassy Heathland and Herbfield* (Williams et al., 2014). Typical fire intervals vary from 50 to 150 years (Zylstra 2006; Williams et al. 2008). with intervals longer in Tasmania (Kirkpatrick & Bridle 2013). High intensity fires are infrequent and usually lit by lightning (Williams et al. 2008). Burning of alpine landscapes by Indigenous people was likely limited, although the frequency and extent of burning in these regions is not fully known (Williams et al. 2008; Zylstra 2018).

Spring vegetation growth and high summer temperatures generates fuel, and areas with higher shrub cover are more flammable relative to dense tussock-forming grasslands or herbfields. Shrubs accumulate larger, more aerated fuel loads (e.g., leaf litter), forming a positive feedback loop between increasing shrub cover and increasing landscape flammability (Williams et al. 2006; Fraser et al. 2016; Camac et al. 2017). Grasses germinate and resprout strongly after fire (hence termed ‘facultive resprouter') (McDougall & Walsh 2007). Dominant shrubs such as *Grevillea australis, Asterolasia trymalioides*, *Phebalium squamulosum*, *Hovea montana* and *Pimelea alpina* and forbs are killed by fire. Post-fire persistence of these species relies on germination from soil and canopy seedbanks (hence termed 'obligate seeders') (McDougall & Walsh 2007). Seeds dispersed through gravity (< 10 m), wind or insect transport (> 100 m) germinate *en-masse* following a fire (Venn et al. 2017; Morgan & Venn 2017). Grasses germinate within a few weeks post-fire and obligate seeder shrubs germinate within a year (Williams et al. 2008). Small populations of fire-tolerant resprouter shrubs also recover post-fire by regenerating their previously burnt canopy or via germination from the seedbank (Camac et al. 2013; Lamont & Keith 2014). Species composition can take up to five years to return to a pre-fire state, although canopy height and the typical low bare ground cover may take more than a decade to recover (Camac et al. 2013; Verrall & Pickering 2019).

Fire is not a characteristic disturbance in Tasmania (Kirkpatrick & Bridle 2013; Venn et al. 2017) . However, *Poa spp.* of *Alpine-subalpine Open Grassy Heathland and Herbfield* exhibit fire-tolerant traits retained from ancestral, temperate species (Kirkpatrick & Bridle 2013; Venn et al. 2017) . A single fire can eliminate other flora in the ecosystem. A single fire may eliminate other fire-sensitive flora in the ecosystem (e.g., occasional shrubs, *Eucalyptus spp*. trees). Recovery of these fire-sensitive flora is reliant on recolonisation bare ground (Kirkpatrick & Bridle 2013; Kitchener & Harris 2013).

Snowpack

Snowpack depth in *Alpine-subalpine Open Grassy Heathland and Herbfield* varies between seasons and regions. Deep, prolonged snowpack formation is typical of alpine extents; prolonged snow is rare in Tasmanian extents and restricted to the highest elevations (Harris & Kitchener 2005; Venn et al. 2017). Snowpack acts as an insulating layer, protecting low-lying vegetation such as herbs, vegetation roots and (if depth is sufficient) dwarf shrubs from frosts, frost heave, and strong winds (Lindenmayer et al. 2014; Venn et al. 2017). The discontinuous nature of the open shrub canopy from adjacent closed heathland precludes formation of a subnivean space. However, snowpack accumulation on shrub canopies may create warmer local microclimates that insulate understorey flora and shelter fauna during winter, including the mountain spotted grasshopper (*Monistria concinna)* and broad-toothed rat (*Mastacomys fuscus*) (Green & Osborne 1994).

Frost heave

Frost heave is a naturally recurring seasonal disturbance that creates bare ground and provides opportunities for vegetation recruitment (Williams et al. 2014). Frost heave occurs when surface and sub-surface water freezes, causing needle ice to form and uplift the soil substrate. These uplifted soils expose roots to freezing, desiccating conditions and eventually cause plants to dieback or senescence. Repeat freeze-thaw cycles during snowmelt or where an insulating snow layer is absent exacerbate frost heave and soil erosion as ice melts. Bare ground is highly susceptible to severe frost heave. The capacity of the soil to insulate plant roots to high temperature events (such as high temperatures or fire) is also reduced, and plant mortality and vegetation recruitment may decline following winter frost heave (Wahren et al. 2013).

Bare ground and recruitment

Bare ground is generally low in *Alpine-subalpine Open Grassy Heathland and Herbfield* (< 5%; Camac et al. 2013; Williams et al. 2014). Bare ground provides recruitment opportunities for grasses shrubs. Without bare ground, shrubs cannot establish and grasses and herbs remain dominant (Williams et al. 2008). Bare ground increases following natural disturbances such as fire and drought. Herbivore grazing, vegetation senescence and frost heave also create bare ground. Bare ground recovers to low levels (< 3 %) within 5-10 years (Costin et al. 1979; Williams & Ashton 1987). Rates of vegetation recovery (and concurrent declines in bare ground cover) may be reduced if the area is exposed to grazing pressure (Bridle & Kirkpatrick 1999).

Major threats

Climate change

Climate change is a major threat to *Alpine-subalpine Open Grassy Heathland and Herbfield*. Declines in precipitation, increasing temperatures and earlier onset of the growing season have been observed since 1979, whilst declines in snowpack duration and depth have been recorded since the 1950s (Hennessy et al. 2008; Sánchez-Bayo & Green 2013; Venn et al. 2017; Ji et al. 2022). Warmer conditions may reduce time to reproductive maturity for alpine flora (Camac et al. 2017) and induce earlier flowering (Green 2010; Venn et al. 2017), which may lead to a mismatch between flower emergence and pollinator migration, reducing the reproductive capacity of flora. Declines may also occur if seasonal herbivory exceed the capacity of flora to reproduce (Green 2010) or recover from a disturbance (Bridle & Kirkpatrick 1999). Warmer conditions will also enable shrubs to establish following disturbance in elevations previously outside of the historical temperature- and late-lying snow-limiting elevational ranges (Auld et al. 2022; Hickman et al. 2024). The subsequent increase in shrub growth may enhance fire frequency, removing non-shrub vegetation, and over the long-term (> 50 years), result in a positive fire-shrub-temperature feedback loop (Camac et al. 2017). As a result, shrub thickening – either via thickening of established shrub species, or increased rates of germination and establishment of new shrubs – may threaten the capacity of *Alpine-subalpine Open Grassy Heathland and Herbfield* to maintain the characteristic composition and structure.

Novel fire regimes

Changes to fire regimes pose a significant threat to *Alpine-subalpine Open Grassy Heathland and Herbfield*. Warmer, drier abiotic conditions across south-eastern Australia have increased extreme fire weather patterns, resulting in more frequent and severe fire events (Pickering 2007; Bradstock et al. 2014). *Alpine-subalpine Open Grassy Heathland and Herbfield* is generally tolerant to fire, yet very short fire intervals (< 5 years) may prevent obligate seeding vegetation from reaching reproductive maturity and establishing a soil or canopy seedbank (McDougall et al. 2015) and increase soil erosion. Short interval fires also exacerbate soil erosion (driving loss of the soil-stored seedbank) and are associated with exotic plant invasion (Williams et al. 2014; Van Klinken & Friedel 2017; McDougall et al. 2018).

Invasive plant species

Exotic plants outcompete native vegetation (McDougall et al. 2018) and persist within the soil seedbank (McDougall et al. 2018). The severity and extent of exotic plant invasion in *Alpine-subalpine Open Grassy Heathland and Herbfield* varies with the amount of bare ground, the proximity to established exotic plant populations and the presence of vectors including tourists, invasive fauna, native macropod herbivores and vehicles (Williams et al. 2014; Van Klinken & Friedel 2017; McDougall et al. 2018). Key exotic species include *Pilosella* spp. (hawkweed), *Leucanthemum vulgare* (oxeye daisy) and *Cytisus scoparis* (Scotch broom) (Venn et al. 2017). *Carex flagillifera* (weeping brown or New Zealand sedge) is an emerging threat to in the Tasmanian distribution (Kitchener & Harris 2013). Climate change is expected to exacerbate exotic plant invasions: warmer temperatures allow longer residence times and earlier upward movement of exotic ungulates (as demonstrated in Norway; Rivrud et al. 2019), enable faster rates of woody tissue growth and plant establishment, and increase recruitment opportunities as bare ground cover increases due to more frequent fires (Venn et al. 2017).

Native shrub encroachment

Native shrubs are characteristic of many vegetation complexes within *Alpine-subalpine Open Grassy Heathland and Herbfield*. Ecosystem persistence is not reliant on retention of the shrub component. However, increasing temperatures are driving in-situ shrub growth and thickening whilst altered disturbance regimes are increasing opportunities to encroach into once range limited extents (Camac et al. 2015, 2017; Fraser et al. 2016; Kirkpatrick & Bridle 2016; Moss 2024). Declines are driven by the formation of a closed shrub canopy: light transmission beneath the canopy is reduced and canopy litterfall smothers understory vegetation (Williams & Ashton 1988). Likewise, warmer temperatures and disturbances are also driving the establishment and growth of *Eucalyptus pauciflora* and thus ecosystem transition into *Subalpine Woodlands and Forests* (Naccarella et al. 2020).

Exotic fauna and herbivory

Warmer conditions and earlier snowfall and snow melt are facilitating earlier upward migration of native (e.g., insects, native fauna) and exotic herbivores. Seasonal herbivory (i.e., migration during growing season and summer) is characteristic of *Alpine-subalpine Open Grassy Heathland* with flora adapted to withstand seasonal grazing pressure from native species (Nimmo & Miller 2007; Schulz et al. 2019). Declines may occur if this grazing pressure (either from exotic and native fauna) exceeds the capacity of flora to recover (Green 2010) particularly following a disturbance such as drought or fire (Bridle & Kirkpatrick 1999).

The low, open canopy of *Alpine sub-alpine Open Grassy Heathland and Herbfield* allows exotic species to easily disperse throughout the landscape (Claridge 2016). Exotic species such as deer (e.g., *Cervus unicolor*), horses (*Equus caballus*), pigs (*Sus scrofa*) and rabbits (*Oryctolagus cuniculus*) compete with native herbivores for food (Nimmo & Miller 2007; Green 2010; Lindenmayer et al. 2014);and cause significant damage to *Alpine-subalpine Open Grassy Heathland and Herbfield* (Green & Osborne 1994; Nimmo & Miller 2007; Claridge 2016). These species compact soils (decreasing soil water infiltration and pore space), and remove soil-stabilising vegetation cover through wallowing, grazing, trampling and digging (Nimmo & Miller 2007). This activity increases erosion and creates bare ground, providing favourable conditions for exotic plants to establish – these exotic seeds dispersed as highly mobile exotic herbivores move throughout the landscape (Williams et al. 2014). Fauna also disperses plant pathogens from lower elevations (e.g., *Phytophthora*) (Rigg et al. 2018).

Livestock grazing (and activities associated with pastoralism, such as fire) has permanently changed the composition of many alpine ecosystems (Zylstra 2006; Moss 2024). *Alpine sub-alpine Open Grassy Heathland and Herbfield* experienced livestock grazing after European colonisation (c. 1840). The use of fire by pastoralists (to promote pasture), preferential grazing by cattle (e.g., *Asterolasia trymaloides* and *Grevillea australis*) and trampling by livestock (i.e., trampling of *Alpine sub-alpine Bogs and Associated Fens*) may have promoted transitions and expansion of this ecosystem type (Zylstra 2006). Grazing ended in 1949 in Kosciuszko National Park (NSW) and in 2003 in Victoria (following the 2003 alpine fires) (Scherrer & Pickering 2005; Williams et al. 2006). Sheep grazing continues on ecosystem extents that occur on private land in Tasmania. Extents of this ecosystem type may represent collapsed shrub and woodland ecosystems (Bridle & Kirkpatrick 1999).

Plant pathogens

Pathogens such as *Phytophthora cinnamomi, P. cambivora,* and *Armillaria* are potential threats to the characteristic shrub species of *Alpine sub-alpine Open Grassy Heathland and Herbfield* (e.g., *Grevillea spp*., *Prosanthera spp.* and *Pheblium spp.*) (McDougall et al. 2015; Rigg et al. 2018).

Dieback due to *Phytophthora* has been recorded in shrub and woodland ecosystems (Barrett & Yates 2015; Rigg et al. 2018) and the disease is exhibiting local cold adaptation (Khaliq et al. 2019). Whilst ecosystem persistence is not reliant shrubs persisting, shrubs shelter understorey flora from the cold during winter and from hot, desiccating temperatures and sun during summer (Ballantyne & Pickering 2015a). Alteration of the shrub canopy will likely induce cascading and network effects (e.g., altered albedo, biochemical cycling, landscape flammability), although there is insufficient data to determine whether this will significantly threaten *Alpine-subalpine Open Grassy Heathland and Herbfield.*

IUCN Stress Classification

1.2 – Ecosystem degradation
1.3 – Indirect ecosystem effects

IUCN Threat Classification

1.3 – Tourism and recreation areas
2.3.1 – Nomadic livestock grazing
7.1.1 – Increase in fire frequency / intensity
8.1 – Invasive non-native species
11.1 – Habitat shifting and alteration
11.2 – Droughts
11.3 – Temperature extremes

Ecosystem collapse

Ecosystem collapse of *Alpine-subalpine Open Grassy Heathland and Herbfield* occurs in several ways. The ecosystem type may transition into *Alpine-subalpine Closed Heath* if shrubs become the dominant structural feature. Likewise, if trees establish, the ecosystem would transition to *Alpine-subalpine Woodland and Forest* with a grassy or open heath understorey. High bare ground (driven by short interval fires or large invasive herbivores) increases soil erosion and likelihood of frost heave: if characteristic grass and herb flora are lost and unable to regenerate from the soil-stored seedbank, this would indicator collapse into a novel, depauperate ecosystem type. Collapse would also occur if herbivory rates exceeded the capacity of flora to recover and establish a seedbank: sheep and rabbit grazing and recurrent fire (from lightning strikes and land clearing) has already caused ecosystem collapse on private land in Tasmania (Bridle & Kirkpatrick 1999; Venn et al. 2017). Finally, theoretical collapse would occur if exotic flora became dominant after a disturbance, although this scenario is unlikely in the near future.

In summary, ecosystem collapse in *Alpine-subalpine Open Grassy Heathland and Herbfield* is this assessment occurs when:

1. *Area*: The mapped distribution declines to zero (100% loss) (Criteria A and B), or;
2. *Fire Frequency:* Fires increase to a frequency of ≥ 1 in every 5 years (range: 5 - 15 years) (Criterion C), or;
3. *Bare Ground:* Cover of bare ground increases to ≥ 30 % (range: 30 – 50 %) five years after a disturbance, with no signs of vegetation recovery (Criterion C), and / or;
4. *Shrub cover:* Cover of shrubs increases to ≥ 70 % (range: 50% - 70 %) (Criterion D).
5. *Tree abundance: Eucalyptus pauciflora* establishes in the ecosystem (Criterion D).

Risk Assessment

Criterion A: Reduction in distribution

Summary

The risk status of *Alpine-subalpine Open Grassy Heathland and Herbfield* is assessed as **Least Concern** under sub-criteria A1, A2, and A3.

Methods

Recent decline (A1)

Maps of present-day extent were constructed from the most accurate and spatially comprehensive spatial data at the time of assessment (archived in 2023). The distribution of mapped vegetation subunits that aligned with *Alpine-subalpine Open Grassy Heathland and Herbfield* was obtained for extents in ACT Government (ACT Government 2018)(1:10000, 100 m with 83% accuracy: Baines et al. 2013), NSW (*Forest Ecosystems: Vegetation of the Southern Forests VIS ID 3858* , 1 : 25000, 250 m to 1 : 100000, 1 km archived at 2005, with reasonable to good accuracy) (Gellie 2005), Victoria (*Native Vegetation – Modelled Extent 2005* (1 : 25000, 250 m, archived in 2005, with good accuracy although recommended scale at 1:100000) (Newell et al. 2006) and Tasmania (*TASVEG 4.0*, 1 : 25000, 2.5 m – 17. 5m, with 90 % accuracy; (Kitchener & Harris 2013; DPIPWE 2020). A proportion of the vegetation community *EVC 44 Sub-alpine Treeless Vegetation* (Victoria) was assigned to this ecosystem type by experts based on high resolution aerial photography. Distributions mapped in 2005 (NSW, Victoria), 2012 (ACT), and between 2017 to 2019 (Tasmania) were assumed representative of present-day extent (2020).

Mapped historical extents were not available at the time of assessment. Thus, we inferred the likelihood of declines occurring due to grazing, tourism, land clearing, drought, fire and evidence of ecosystem decline or expansions from published scientific literature and theses.

Future change (A2)

We used patterns in observed responses in vegetation to disturbances (fire and temperature change) from observation and experimental field studies to project likely future change (Camac et al. 2013, 2017). We also used expert estimates of future changes in shrub, grass, herb and woodland flora in grassland and heathland ecosystems from Camac et al. (2021); data included 22 expert estimates of the current (2017) and likely future (2050) cover of characteristic species including *Grevillea australis*, *Poa heimata*, *Celmisia spp*. and *Craspedia spp*.

Historical change (A3)

Modelled maps of historical distribution at 1750 were available for NSW (*Forest Ecosystems: Native Vegetation of the Southern Forests: South-east Highlands, Australian Alps, South-west Slopes, and SE Corner bioregions. Pre 1750. VIS ID 3859*) and Victoria (*Native Vegetation – Modelled 1750 Ecological Vegetation*). Modelled ACT and Tasmanian distributions were unavailable at the time of assessment; extant vegetation was added to NSW and Victorian maps to estimate ecosystem distribution in 1750.

To account for climate change, two historical scenarios were modelled to estimate changes in vegetation distribution: Scenario 1 (conservative estimate of 1750 distribution) assumed that the elevational range of vegetation communities remained stable over time; the same elevation cut-off used to develop present-day ecosystem maps was assigned as the minimum elevation of the ecosystem type at 1750. Scenario 2 (a non-conservative estimate) incorporated encroachment of shrubs from lower elevation ecosystems (assuming that average temperatures have increased at a steady rate since 1950, the elevational range of shrubs increased 50 m since 1750) (Hennessy et al. 2008; Slatyer 2010). To assess sub-criterion A3, the present-day distribution was compared to the modelled distribution in 1750 from Scenario 1 and Scenario 2, and the percentage change used to determine the risk outcome.

Assessment outcome

Recent change (A1)

Historical livestock grazing and land clearing of *Alpine-subalpine Open Grassy Heathland and Herbfield* comments throughout mainland and Tasmanian extents in the late 19th Century (c. 1820 – 1840) (Zylstra 2006; Kirkpatrick & Bridle 2013). Extensive livestock grazing, and the use of fire to clear land for pasture occurred throughout the mainland extents: grazing declined c. 1949 following the establishment of Kosciuszko National Park in NSW, followed by a ban in the early 1960s (McDougall & Walsh 2007). Grazing continued in Victoria, although the use of fire to clear land declined during the 1990s; grazing ceased in 2003 following extensive alpine fires and was banned in alpine Victoria in 2005 / 2006. Following the release of mainland extents from grazing, substantial increases in shrub cover were observed (Scherrer & Pickering 2005; Williams et al. 2006; Camac et al. 2015). The use of fire to clear land for grazing in Tasmania declined during the 1980s (Kirkpatrick & Bridle 2013): private land grazing continues in small extents on the Central Eastern Plateau, although this likely represents a small portion of ecosystem distribution. Although more recent evidence of ecosystem condition is limited (Bridle & Kirkpatrick 1999; Kirkpatrick & Bridle 2016), it is likely that prolonged grazing and maintains the persistence of derived *Alpine-subalpine Open Grassy Heathland and Herbfield* by supressing shrub and tree establishment from historically tree and shrub-dominant ecosystem types.

Evidence from aerial photography shows that ecosystem distribution on the mainland has fluctuated through time, with existing extents of *Alpine-subalpine Closed Heath* expanding into *Alpine-subalpine Open Grassy Heathland and Herbfield* (McDougall 2003). Minor areas of *Alpine-subalpine Open Grassy Heathland and Herbfield* continue to be cleared for ski runs at Falls Creek, Victoria and Mt Hotham, Victoria (McDougall & Walsh 2007), however, the total 'skiable' area is < 3 km^2, or < 1 % of the total ecosystem extent. More recently, declines in the extent of *Snowpatch Herbfields* ecosystems have been driven by expansion of Tall Alpine Herbfield communities associated with *Alpine-subalpine Open Grassy Heathland and Herbfield* (Morgan & Walker 2023). Assuming that trends observed on the mainland are representative of those in Tasmania, any declines in ecosystem distribution over the past 50 years have been minor and likely offset by ecosystem expansion. Thus, the ecosystem type is classified as **Least Concern** under sub-criterion A1.

Historical livestock grazing and land clearing (c. 1850) of *Alpine-subalpine Open Grassy Heathland and Herbfield* across the Bogong High Plains, Victoria, ceased in 2003 (Williams et al. 2006; McDougall & Walsh 2007) and in 1949 in Kosciuszko National Park, NSW. Increasing rates of shrub cover were observed following the end of grazing in these regions (Scherrer & Pickering 2005). Extents of grassland transited into open heathland, and open heathland collapsed into *Alpine* *sub-alpine Closed Heath* following the end of grazing (McDougall 2003). The cover of *Alpine sub-alpine Closed Heath*, wetlands and trees (*Eucalyptus pauciflora*) increased during this period, likely representing recovery and expansion of existing patches rather than emergence of new patches.

Minor areas of *Alpine-subalpine Open Grassy Heathland and Herbfield* continue to be cleared for ski runs at Falls Creek and Mt Hotham, Victoria (McDougall & Walsh 2007), however, the total 'skiable' area is < 35 km^2 (Sato et al. 2014; MountainWatch 2023a, 2023b) or < 5 % of the total ecosystem extent. The area of hiking tracks, roads and infrastructure throughout the ecosystem type is not known. Private land grazing continues across small parts of the Tasmanian distribution, although more recent evidence of ecosystem condition is limited (Bridle & Kirkpatrick 1999). Likely, prolonged grazing and maintains the persistence of derived *Alpine-subalpine Open Grassy Heathland and Herbfield* by supressing shrub and tree establishment. Due to the minor changes in the distribution over the past 50 years, the ecosystem type is classified as **Least Concern** under sub-criterion A1.

Future change (A2b) (2005 - 2045)

We anticipate that the distribution of *Alpine-subalpine Open Grassy Heathland and Herbfield* will likely fluctuate at local scales. Increasing fire frequency, drought and higher summer temperatures may drive widespread mortality events of grass and herb flora (Grose et al. 2010; Griffin & Hoffmann 2012; Ji et al. 2022). The distribution of mainland *Alpine-subalpine Open Grassy Heathland and Herbfield* may decline if in-situ shrubs thicken and shrubs and trees encroach from adjacent *Alpine-subalpine Closed Heath* and *Alpine-subalpine Woodland and* Forest. This is most likely to occur after a disturbance event as establishing trees and shrubs colonise bare ground colonising bare ground from these disturbance events (Camac et al. 2017; Verrall 2023; Hickman et al. 2024). Expert estimates anticipate an increase in the cover and upper elevational range of trees and shrubs by 2050 (Camac et al. 2021) although declines in other ecosystem types (e.g., *Snowpatch Herbfield*) (Morgan & Walker 2023) driven by encroachment of grasses and herbs of *Alpine-subalpine Open Grassy Heathland and Herbfield* may offset these declines.

Evidence available to project likely changes in Tasmanian extents is limited. Tasmanian alpine climates are moderated by coastal proximity rather than elevation as on the mainland and these maritime climates may mitigate the effect of global warming (Grose et al. 2010; Henríquez et al. 2023; Kirkpatrick et al. 2024). As flora of many Tasmanian alpine ecosystems lack tolerance to fire, fire disturbances may provide opportunities for fire-tolerant *Poa spp.* of *Alpine-subalpine Open Grassy Heathland and Herbfield* to colonise burnt ecosystems and thus expand (Foulkes et al. 2021). Conversely, continued fire suppression may drive shrub and tree encroachment in present-day Tasmanian extents of *Alpine-subalpine Open Grassy Heathland and Herbfield* (Kirkpatrick & Bridle 2016; Foulkes et al. 2021). Any changes to ecosystem distribution are likely to be minor in the near future and the ecosystem is considered **Least Concern** under sub-criterion A2b. However, we caution that long-term (c. 2100) transitions to *Alpine-subalpine Closed Heath* are likely. Warmer temperatures, reduced snowfall and increased disturbance frequencies are already driving regional declines in grasslands alongside the expansion of trees and shrubs (Verrall 2023) and these signals are subject to time lags.

Historical change (A3)

The estimated change in ecosystem distribution between 1750 and present day was less than 3 %. The area in 1750 was estimated to be 573.7 - 574.1 km^2 for Scenarios 1 and 2 respectively, with the current distribution estimated at 571.0 - 571.4 km^2. Therefore, this ecosystem is considered **Least Concern** under sub-criterion A3.

Criterion B: Restricted distribution

Summary

Alpine-subalpine Open Grassy Heathland and Herbfield is assessed as **Least Concern** under all sub-criteria.

Methods

The Extent of Occurrence (EOO, sub-criterion B1) and Area of Occupancy (AOO, sub-criterion B2) of *Alpine-subalpine Open Grassy Heathland and Herbfield* were determined using a combination of existing map products from the ACT (ACT Government 2018), New South Wales (DCCEEW 2011a), Victoria (DELWP 2021) and Tasmania (Harris & Kitchener 2005; DPIPWE 2020), with data for subalpine treeless Victoria augmented by experts with high resolution aerial photography from 2011 to better reflect on-ground ecosystem boundaries. Estimates of EOO and AOO were calculated using the R package, *Redlistr* (Lee et al. 2019). The number of threat-defined locations (sub-criterion B3) was based on fire as this is the most important known stochastic threat to the ecosystem type.

Assessment outcome

Sub-criterion B1

The Extent of Occurrence (EOO) for *Alpine-subalpine Open Grassy Heathland and Herbfield* is 146,760.22 km^2 (Figure 20), equating to a risk status of **Least Concern** under sub-criterion B1.

Sub-criterion B1

The ecosystem type occupies 177 10 x 10 km grid cells and is therefore **Least Concern** under sub-criterion B2.

Sub-criterion B3

The major stochastic threat for *Alpine-subalpine Open Grassy Heathland and Herbfield* is fire. The ecosystem type is resilient to large scale, intense, but infrequent fires. Between 1970, and 2020, fires have burnt the ecosystem type in 43 fire seasons (from July to June), covering 427.5 km^2 (~69% of the distribution). Most fires have only burnt a small portion of the distribution, although several large fires have burnt extensive areas (fire seasons: 1938/1939, 2002/2003, 2006/2007, 2019/2020). In 2003, large-scale fires burnt > 50% alpine vegetation at Kosciuszko National Park, NSW (McDougall et al. 2015) and ~ 50% at Bogong High Plains, Victoria. However, *Alpine-subalpine Open Grassy Heathland and Herbfield* vegetation is resilient to infrequent fires and the ecosystem type is patchily spread over an extensive region. Whilst large fires may spread, not all extents are likely to be burnt during the same fire, or burnt to the same severity, and the whole ecosystem type is unlikely to be burnt multiple times within a very short timeframe (< 5-year interval) sufficient to cause collapse within 20 years. Further, the ecosystem is distributed across > 5 threat-defined locations as it is widespread and patchy. Therefore, the ecosystem type is considered **Least Concern** under sub-criterion B3.

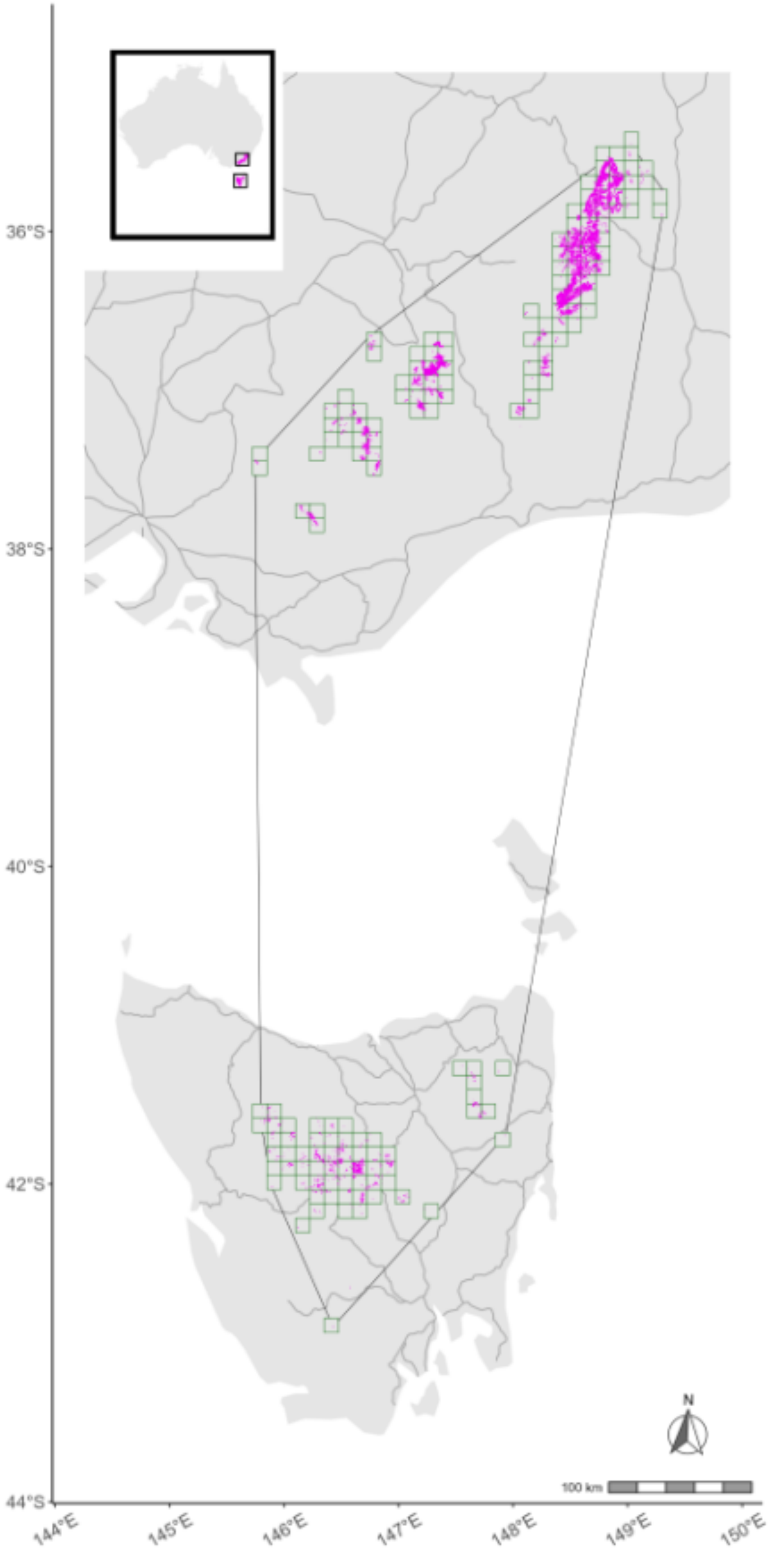


Figure 20. Map of *Alpine-subalpine Open Grassy Heathland and Herbfield* (magenta polygons) across Australia, showing Extent of Occurrence (black polygon) and Area of Occupancy where the 1% rule was not applied (green squares).

Criterion C: Environmental degradation

Summary

The ecosystem type is assessed as **Least Concern** under sub-criterion C1 and C2, and Data Deficient under sub-criterion C3.

Identification of abiotic indicators

We considered two indicators to assess risk of collapse by environmental degradation:

- *Fire Frequency:* a measure of the frequency in which the same area burns.
- *Bare ground*: a measure of groundcover lacking vegetation, but including leaf litter, rocks, and bare soil.

Indicator: fire frequency (%)

Relevance to ecosystem function

Fire is a characteristic process in *Alpine-subalpine Open Grassy Heathland and Herbfield*, facilitating the recruitment of grasses, forbs, herbs and obligate seeding shrubs (Zylstra 2018). Historically, fires were an infrequent disturbance event in alpine landscapes, only occurring every 50 - 150 years (Williams et al. 2008, 2014). However, fire frequency has increased since the 1990s in Tasmania (Kirkpatrick & Bridle 2013) and since 1987 in Victoria (Tran et al. 2020). Climate change is expected to continue increasing the frequency and severity of alpine fires (Pickering 2007). Repeated, short fire intervals represent a threat to *Alpine-subalpine Open Grassy Heathland and Herbfield*. On the mainland, short fire intervals alter species composition (Camac et al. 2013). Collapse occurs if the capacity of the ecosystem to recover from fire is lost or if non-characteristic flora recruit into bare ground extents (e.g., shrubs, trees, invasive exotic flora) such that grasses and herbs are no longer the defining feature of the ecosystem type.

Data availability and quality

Recent change (C1) and Historical change (C3)

Historical records of bushfires and planned burns are available as shapefiles, detailed in the main methods.

Future change (C2a)

We used FROST to create 100 replicate simulations of the 20-year period from 2060 to 2079 (Mccoll-Gausden et al. 2022). For each simulation, we extracted a spatial raster that displayed the number of fires burnt within the 20-year period, for each 180 m^2 cell. For each climate scenario, we calculated the percentage of the ecosystem type (Extent, based on the number of cells) that burnt at a frequency that exceeded the collapse threshold (assuming Relative Severity [RS] = 100%).

Selection of collapse threshold

For mainland extents of the ecosystem type, fire frequencies of $<$ 5 years (5 - 15 years) between fires with no signs of regeneration of the characteristic vegetation would indicate ecosystem collapse. The collapse threshold of $\geq$ 1 fire in 5 years equates to a spatially weighted annual probability of burning of 0.20.

Selection of initial and present/future values

Recent change (C1)

To define the initial and present fire frequency, we analysed the fire history data sets of (DSE 2020b; NSW Goverment 2022; Forest Fire Management Victoria 2023; Tasmanian Government 2024). Maps of ecosystem extent were overlaid with fire history datasets from government sources to calculate the proportion of ecosystem extent that experienced repeat burns less than the collapse threshold to determine the spatially weighted annual probability of decline (pSW). To generate a timeseries, we calculated the number of times and area burnt between 1970 - 1979 (initial value), 2010 - 2020 (current value), and at 10-yearly intervals between these timeframes. We then compared pSW at initial and final timeframes relative to the collapse threshold to calculate the RS of decline (following IUCN 2024, p. 61: RS = ((pSW (1970 - 1980) - pSW (2010 – 2020)) / (pSW (2010 – 2020) - pSW (Collapse))) × 100).

Future change (C2)

The risk of future declines due to short fire intervals (Sub-criterion C2a) were assessed using 'Fire Regime and Operations Tool' (FROST) models (Mccoll-Gausden et al. 2022) (FLARE Wildfire Research, https://www.flarewildfire.com/software/frost-family/) with 100 future fire models constructed considering weather, fuel and biomass accumulation under four Representative Concentration Pathways (RCP) scenarios (CSIRO R1, CSIRO R3, ECHAM R1 and ECHAM R3). For each replicate, we extracted the spatial raster of each simulation and counted the number of times each 180 m2 cell was burnt between 2060 to 2079. We then counted the number of grid cells, and thus Extent (%) of each ecosystem burnt at intervals equal to or exceeding the collapse threshold (assuming RS = 100 %).

Historic change (C3)

We used the same approach as used in sub-criterion C1 to calculate initial and present values. In lieu of historical data, we assumed that the fire frequency in 1939 - 1949 was relatively representative of the fire frequency c. 1750.

Calculation of severity and extent

Recent change (C1)

Fires have burnt *Alpine-subalpine Open Grassy Heathland and Herbfield* in 43 fire seasons between 1970 and 2020 (Table 10), covering a total of ~ 427.5 km^2 (~69 % of the ecosystem type.) None of the area burnt at or above the collapse threshold (≥ 1 fire every 5 years) in any period (Table 10). Most of the ecosystem type was burnt only once in the 50-year period. The area burnt in each 10-year period has increased, from 4.65% in 1970-1980, to 25.18% in 2020. This equates to an increase from 0.0046 to 0.025 in the spatially weighted annual probability of burning. Thus, the relative severity is 10.44 % (((0.0046 - 0.025) / (0.0046 - 0.2)) × 100). Thus, given the there were no areas with fires at frequencies above the collapse threshold, and in combination with a low increase in relative severity, the risk status is **Least Concern** under sub-criterion C1.

Table 10. The percentage (%) of the *Alpine-subalpine Open Grassy Heathland and Herbfield* area (based on the number of cells) that have burnt over the threshold from 1970 - 1980, 1980 - 1990, 1990 - 2000, 2010 - 2020 and 2000 – 2020 and the actual respective fire frequencies.

Period of 10-year interval	**Percentage of ecosystem burnt over threshold (1/5 years)**	**Number of fires**	**Percentage of ecosystem burnt**	**Spatially weight probability of burning**
1970-1980	0%	1	4.65%	0.46
1980-1990	0%	1	3.96%	0.39
1990-2000	0%	1	1.99%	0.19
2000-2010	0%	1	44.99%	4.4
2010-2020	0%	1	25.18%	2.5

Future change (C2)

Under all four climate scenarios, there were low predicted fire frequencies and most of the mainland extent was not predicted to burn at a frequency that exceeded the collapse threshold (Table 11). The mean predicted fire frequency was less than the collapse threshold for the entire mainland extent (Figure 21). The highest fire frequency predicted (one fire in 20 years) was in the most northern part of the ecosystem type extent (Figure 21). Overall, the probability of any part of the mainland distribution (i.e., at least one grid cell) burning at a frequency that met or exceeded the collapse threshold ranged from 2% (CSIRO - R1) to 4 % (CSIRO - R3, ECHAM - R1, ECHAM R3). However, the extent of the ecosystem type that exceeded the collapse threshold (i.e., relative severity = 100%) in one or more simulations ranged from 0.25 % (CSIRO-R1) to 3.59 % (ECHAM-R1). Very small parts of the mainland extent displayed a very low relative severity, and this is likely to also be representative of the Tasmanian distribution. However, evidence from literature suggests that warmer, drier conditions are more conducive to fire are likely by 2050, with substantial increases in the magnitude and duration of conditions conducive to fire (Clarke et al. 2011; Ji et al. 2022). Studies of interacting temperature, precipitation and post-fire recovery also suggest that flora may respond to fire in non-linear, multidirectional ways that are difficult to predict (Hickman et al. 2024). Therefore, the ecosystem type is **Least Concern** under sub-criterion C2.

Table 11. The percentage (%) of *Alpine-subalpine Open Grassy Heathland and Herbfield* area (based on the number of grid cells) and the number of simulations (out of 100) that burnt at a frequency that met or exceeded the collapse threshold (≥ 1 fire every 5 years) for each climate scenario.

Number of simulations (/100)	**CSIRO - R1**	**CSIRO - R3**	**ECHAM - R1**	**ECHAM - R2**
0	98.75	97.60	96.41	96.51
1	-	-	-	-
2	0.25	2.38	3.57	3.47
3	-	-	-	-
4	-	0.02%	0.02	0.02

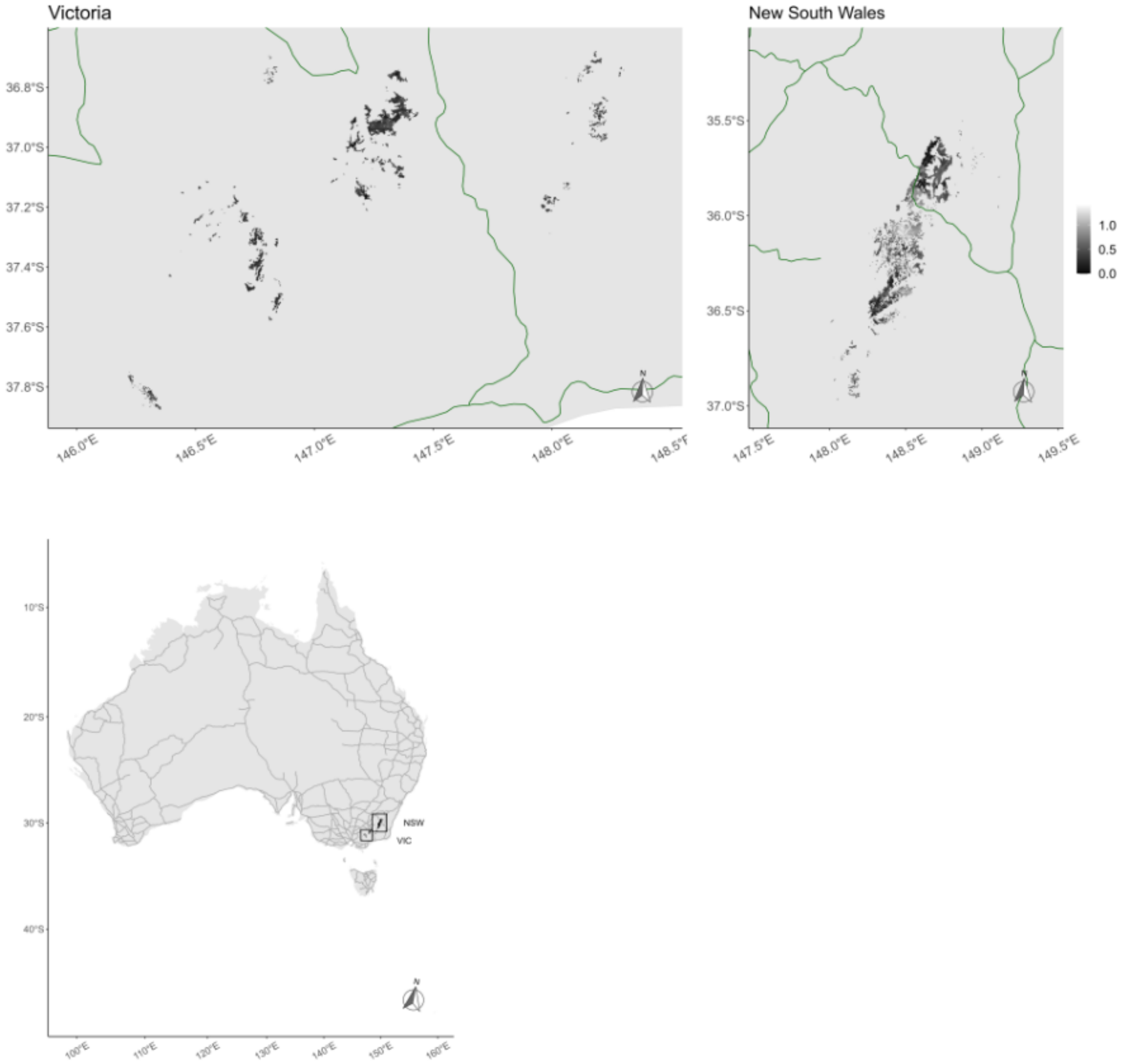


Figure 21. Mean projected future fire frequency in 2060 - 2079, calculated across 100 replicated simulations.

Historical change (C3)

According to fire records, very little of the ecosystem type (< 1%) was burnt between 1939 - 1960 (Table 12). This compares to current records, which suggest 70.1% of the ecosystem type has burnt at least once since 2000. However, fire records prior to 1960 are unreliable and often anecdotal in nature (e.g., Zylstra 2006). The large transformation in area burnt cannot be reliably attributed to changes in fire regimes alone. Therefore, the risk status under sub-criterion C3 is assessed as **Data Deficient.**

Table 12. The percentage (%) of the *Alpine-subalpine Open Grassy Heathland and Herbfield* area (based on the number of grid cells) that has burnt at different frequencies.

Fire frequency	**Current (2000 - 2020)**	**Baseline (1939 - 1960)**
1	70.1	0.22

Indicator: bare ground (%)

Relevance to ecosystem function

Up to 5 % bare ground is characteristic of this ecosystem type (Williams et al. 2014). High levels of bare ground exacerbate the risk of soil erosion, frost heave (Wahren et al. 1994; Williams et al. 2008) and loss of the soil seedbank, with soil erosion a particular threat to Tasmanian extents (Venn et al. 2017). The loss of native plants and the soil seedbank also provides recruitment opportunities for invasive plants. Disturbances such as fire, invasive herbivore grazing, trampling and wallowing by invasive herbivores can increase the amount of bare ground (Bridle & Kirkpatrick 1999; Williams et al. 2014; Camac et al. 2015).

Data availability and quality

Recent change (C1)

Site-specific values of bare ground cover (%) at Gungartan and Kosciuszko National Park in NSW, the Bogong High Plains in Victoria, and the East Central Plateau and Mt Field in Tasmania were sourced from published literature (see Table 13) (Kirkpatrick & Dickinson 1984; Bridle & Kirkpatrick 1999; Scherrer & Pickering 2005; Wahren et al. 2013). Values were only extracted from observational and experimental (control only) field sites verified as *Alpine-subalpine Open Grassy Heathland and Herbfield*. Data were unavailable for other extents of the ecosystem type, including for the ACT. We assumed site-specific data from each location was broadly representative of bare ground in the respective state, and that the NSW data (Kosciuszko National Park and Gungartan) were representative of the distribution in NSW and the ACT.

Future change (C2a) and Historical change (C3)

Datasets measuring historical change and predicting future changes in bare ground were unavailable. The assessment of these sub-criteria was based on experts' inferences and known threat histories across the region.

Table 13. Values of bare ground cover (%) obtained from scientific literature and used to assess sub-criterion C1 of *Alpine-subalpine Open Grassy Heathland and Herbfield*. Values of bare ground were only included if sampling site could be verified as *Alpine-subalpine Open Grassy Heathland and Herbfield*.

Citation	**Location**	**Sampling year**	**Bare ground cover (%)**	**Collapse threshold met?**	**Confounding effects**
(Scherrer & Pickering 2005)	Kosciuszko National Park, NSW	1959	3.9	No	Grazing 1850s - 1949. Drought 1961 - 1968.
		1964	2.7	No	
		1968	5	No	
		1974	2.6	No	
		2001	2.1	No	
	Gungartan, NSW	1959	31.3	Yes: site recovering from grazing (ceased 1949)	Grazing 1850s - 1949. Drought 1961 - 1968.
		1964	24.2	No	
		1968	31.5	No	
		1974	16.5	No	
		2001	12.3	No	
(Williams & Ashton 1987)	Bogong High Plains, Victoria	1980	3	No	No history of livestock grazing
		1983	2	No	
		1980	2	No	Livestock grazing
		1983	9	No	
(Camac et al. 2013)	Bogong High Plains, Victoria	2008	22.6	No	Burnt 2003
		2008	2	No	Unburnt
(Wahren et al. 2013)	Bogong High Plains, Victoria	2004	4	No	Livestock grazing ceased 2002
		2010	3	No	
(Bridle & Kirkpatrick 1999)	Liawnee Moor, East Central Plateau, Tasmania.	1996	10.5	No	Previous grazing, fire c. 1960
		1997	10	No	

(Bridle & Kirkpatrick 1999)	Liawnee Moor, East Central, Tasmania.	1974	0.39	Possibly: heavy grazing (sheep, rabbits) and fire ceased 1960s. Bare ground may have been > 30% prior to 1974.	Previous fire c. 1960
		1993	0.22	No	
(Kirkpatrick & Dickinson 1984)	Mount Field, Tasmania	1969	0.182	No	Previous fire c. 1960
		1969	0.028	No	

Selection of collapse threshold

Bare ground cover is generally low (< 5 %) in undisturbed *Alpine-subalpine Open Grassy Heathland and Herbfield* (Camac et al. 2013; Williams et al. 2014). Following a drought or fire, bare ground cover initially remains high, with vegetation recovery occurring within 5 years (5 - 10 years) (Costin et al. 1979; Williams & Ashton 1987); recovery may be delayed if vegetation is exposed to invasive species (Bridle & Kirkpatrick 1999). We therefore assumed that the ecosystem type would collapse if bare ground was ≥ 30 % (30 – 50 %) without signs of vegetation regeneration 5 years (5 – 10 years) after a disturbance (McDougall et al. 2015).

Shape of decline

In the absence of more comprehensive information, we assumed linear relationships between bare ground cover and progress towards collapse.

Calculation of severity and extent

Recent change (C1)

Evidence suggests that bare ground cover tends to increase after drought or fire due to the removal of leaf litter layer and vegetation. However, most study locations observed bare ground cover of < 30 % five years after a disturbance (Table 13). Bare ground at Liawnee Moor, East Central Plateau in Tasmania exceeded the collapse threshold. The duration (> 5 years) of bare ground cover prior to 1974 could not be determined, although we assume bare ground cover was higher due to agricultural land use (fire for land clearing and heavy sheep grazing ceased in 1960). Vegetation recovered at ~1% per year in subsequent monitoring years (1974 – 1993) (Williams & Ashton 1987) although the site is subject to ongoing rabbit grazing. We assumed that vegetation recovery (and rabbit grazing) was also occurring prior to 1974, and that regeneration is the main mechanism of declines in bare ground cover. Whilst a small portion of ecosystem extent (< 1% of total ecosystem type) at Laiwnee Moor, East Central Plateau, Tasmania may have experienced localised collapse and recovery, bare ground cover across the entire ecosystem type is unlikely to meet the threshold for Vulnerable. Therefore, therefore, the risk status is **Least Concern** for sub-criterion C1.

Future change (C2a)

Bare ground cover may decrease as warmer climates drive faster rates of shrub growth and expansion (Armstrong et al. 2013; Wahren et al. 2013). Conversely, bare ground may increase due to persistent and ongoing threats from introduced herbivores, invasive plants, and land clearing. More frequent fires due to warmer, drier conditions and increased presence of flammable shrub cover (see sub-criterion D1and D2) may increase bare ground in the short term (Wahren et al. 2013; Camac et al. 2017). However, this will likely be counteracted by enhanced rates of vegetation recovery (particularly shrubs) (Williams et al. 2014). In the absence of more comprehensive evidence, we estimate a risk status of **Least Concern** under sub-criterion C2a.

Historical change (C3)

In the absence of land use change and increasing fire frequency, we expect that bare ground cover was < 5 % in 1750 (Williams & Ashton 1987; Scherrer & Pickering 2005; Zylstra 2006). Persistent bare ground cover > 3 % is very rare across the Australian alpine region (pers. comm James Camac). Disturbances that create bare ground in *Alpine-subalpine Open Grassy Heathland and Herbfield* have increased since 1750. Trampling and grazing by livestock and feral ungulates increases bare ground cover on a very local scale (Williams et al. 2014), however the ecosystem type can revegetate following the removal and exclusion of these introduced species (Williams & Ashton 1987; Scherrer & Pickering 2005). Several fires have burnt the ecosystem type, resulting in higher bare ground cover. Recovery typically occurs within a decade on the mainland (Williams et al. 2014), although substantially slower in Tasmania (Kirkpatrick & Dickinson 1984; Bridle & Kirkpatrick 1999). These short-term increases in bare ground are unlikely to be persistent (> 5 years) or widespread (> 30 % of the distribution), and thus the ecosystem type is unlikely to meet the threshold for Vulnerable. We therefore assign the risk status of **Least Concern** under sub-criterion C3.

Indicator: Climate and precipitation-based indicators

Relevance to ecosystem function

Climate change is likely to impact *Alpine-subalpine Open Grassy Heathland and Herbfield,* however the exact impacts are unclear. Precipitation provides an important source of groundwater recharge and snowpack insulates the vegetation, soil, and soil fauna from freezing during winter (Green & Osborne 1994). Climate change is likely to decrease snowfall, induce earlier snowmelt, and increase frost in the absence of a protective snow cover (Sánchez-Bayo & Green 2013). This may exacerbate erosion, soil freezing, and frost heave. In contrast, warmer temperatures coupled with declines in snow coverage, persistence, and depth may promote shrub establishment and increase encroachment by *Alpine-subalpine Closed Heath.*

Data availability and quality

There is substantial experimental and modelled data linking climate change with altered vegetation composition and vegetation change. Long term monitoring programs, such as the International Tundra Experiment (ITEX), use small-scale in-field experimental studies to characterise vegetation response to climate change, at the species level and may be a valuable source of evidence to inform future assessments (see Hoffmann et al. 2010 and Camac et al. 2017 for examples of ITEX experiments in *Alpine-subalpine Open Grassy Heathland and Herbfield*). However, these types of data represent an evidence base largely restricted to a few geographic regions. Quantitative thresholds still need to be determined for climatic indicators for this ecosystem type. Uncertainty also needs to be resolved in how these climate variables influence other abiotic (e.g., soil water) and biotic (e.g., vegetation dynamics) characteristics of the ecosystem type, and how *in turn* these characteristics influence climate at a localised scale.

Criterion D: Disruption of biotic processes and/or interactions

Summary

The risk status of *Alpine-subalpine Open Grassy Heathland and Herbfield* is assessed as **Least Concern (Least Concern – Near Threatened)** for sub-criterion D1 and **Least Concern** for sub-criterion D2 and D3.

Identification of biotic indicators

We examined the relevance and data availability/quality for one indicator to assess the risk of collapse from disruption of biotic processes and/or interactions:

- *Shrub cover:* the percentage of overlapping cover of woody shrub vegetation 0.5 - 2.0 m high.

Indicator: Shrub cover

Relevance to ecosystem function

Alpine-subalpine Open Grassy Heathland and Herbfield consists of a mosaic of grasses, forbs, sedges, and herbs with limited (if any) small-statured shrubs (Camac et al. 2013; Williams et al. 2014). However, warmer climates, severe droughts (Griffin & Hoffmann 2012; Williams et al. 2014) and more frequent fires (Pickering 2007; Bradstock et al. 2014) are likely to increase shrub recruitment and thickening, particularly in grassland and heathland ecosystems (Camac et al. 2021). However, warmer climates, severe droughts (Griffin & Hoffmann 2012; Williams et al. 2014) more frequent fires (Pickering 2007; Bradstock et al. 2014) are likely to increase shrub recruitment and thickening: climate change is already driving transitions of alpine grassland and herbfields into heathland ecosystems in Arctic and sub-Arctic regions (Myers-Smith et al. 2011). If shrub cover increases to > 50 % cover, the composition and structure of the ecosystem type is also likely to change. Shrubs, now the defining characteristic flora are likely to be taller (0.5 - 2.0 m) than the typical dwarf and prostrate shrubs that occur in *Alpine-subalpine Open Grassy Heathland and Herbfield.* Shrubs (more flammable than grass and herb flora) will also alter the flammability of the alpine region. Formation of a connected, closed shrub canopy would allow snow to accumulate on the canopy and create subnivean space underneath, allowing animals such as the broad-toothed rat (*Mastacomys fuscus*) to persist in the landscape during winter. Seasonal migrants may also be affected as grass is replaced by shrubs: population declines would be expected in both migratory herbivores (e.g., *Kosciuscola* grasshoppers) and the predators that rely on a seasonal abundance of prey during the spring and summer breeding seasons (e.g., raptors, insectivorous birds). Declines in pollinator populations (largely herbivorous) would also reduce the volume of viable seedbank produced by grass and herb flora.

Data availability and quality

Recent change (D1)

Limited data on shrub cover in the alpine region could be confirmed as part of the *Alpine-subalpine Open Grassy Heathland and Herbfield.* Data was limited of extents in Victoria (15.5 % ecosystem extent) and Tasmania (9.5 % ecosystem), but supported trends observed from long-term studies in New South Wales (75 % of ecosystem extent). We used data from a permanent photo-quadrat analysis at Gungartan and Kosciuszko National Park, and field studies from the Bogong High Plains

(Victoria) and Liawnee Moor (Central Eastern Plateau, Tasmania) (Bridle & Kirkpatrick 1999; Scherrer & Pickering 2005; Wahren et al. 2013) (Table 14).

Table 14. Values used to assess the indicator of shrub cover (%) in *Alpine-subalpine Open Grassy Heathland and Herbfield*. Values shown in shrub cover (%) represent the mean % ± standard error reported in each study.

Year	**Shrub cover (%)**	**N**	**Details**	**Citation**
1959	3.9 ± 3.7	10	Permanent photo-quadrat in Kosciuszko National Park, NSW	(Scherrer & Pickering 2005)
1964	2.5 ± 2.4	12		
1968	5.5 ± 4.3	10		
1978	6.2 ± 4.5	12		
2001	17.4 ± 9.4	12		
1959	0.1 ± 0.1	17	Permanent photo-quadrats in Gungartan, NSW	
1964	7.1 ± 3.2	18		
1968	8.1 ± 3.1	18		
1978	10.5 ± 4.7	18		
2001	21.7 ± 8.4	17		
2004	5 ± 1.4	13	1×1 m field control plot in Bogong High Plains, Victoria. Sites were unburnt and previously grazed by livestock.	(Wahren et al. 2013)
2010	11 ± 4.5	13		
1996	37.4	60	0.5×0.5 m plots in Liawnee Moor, Central Eastern Plateau, Tasmania. Measure is for tall shrubs. Ungrazed sites.	(Bridle & Kirkpatrick 1999)
1997	37.4	60		
1996	27.2	60	Same as above on rabbit and native grazed sites.	
1997	29.6	60		
1996	25.9	60	Same as above on sheep grazed sites.	
1997	27.8	60		

Future change (D2b)

We used expert estimates of predicted future changes in cover of diagnostic shrub and grass species from Camac et al. (2021). These data include 22 expert's estimates of the current (2017) and likely future (2050) cover of key species within *Alpine-subalpine Open Grassy Heathland and Herbfield*, including *Grevillea australis* and *Poa hiemata*. These estimates were aggregated and presented as means with 95% confidence intervals.

Historical change (D3)

We inferred change since 1750 based on studies on shrub cover over the last century (see sub-criterion D1) and perceptions of experts in the ecosystem type.

Selection of collapse threshold

Shrub cover was low in *Alpine-subalpine Open Grassy Heathland and Herbfield* (< 30%, although shrub cover up to 50% is considered 'open heathland' in Williams et al. 2014). Therefore, we defined the collapse threshold as an increase shrub cover to ≥ 70% (range: 50-70%; Williams et al. 2014), which represents a transition to *Alpine-subalpine Closed Heath*.

Selection of initial and present/future values

Recent change (D1)

Based on Scherrer & Pickering (2005), we used the shrub cover from 1968 to represent the values in 1970 and values for 2001 as a minimum cover estimate for present values. These data are the longest timeseries available and assumed to be representative of the NSW extent (Figure 22).

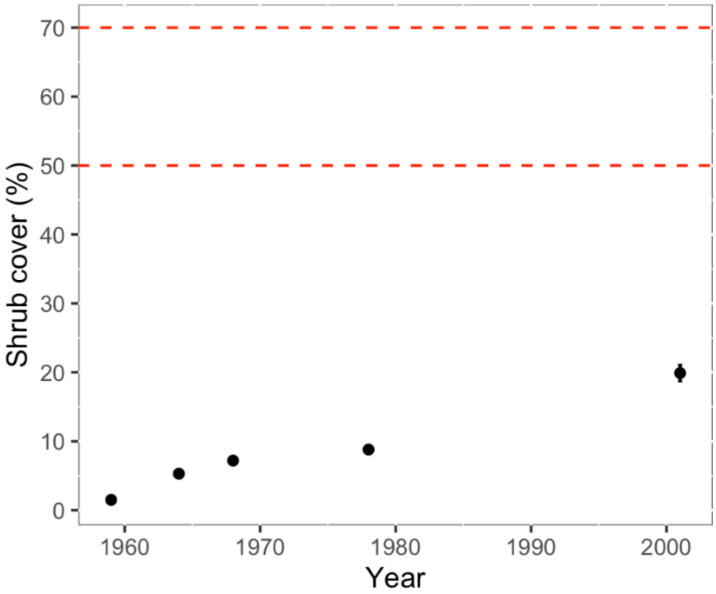


Figure 22. Mean values of shrub cover (%) recorded from field monitoring of *Alpine-subalpine Open Grassy Heathland and Herbfield* in Kosciuszko National Park and Gungartan (NSW) as reported in Scherrer & Pickering (2005). Solid points represent mean values of shrub cover. Vertical lines represent 95% confidence interval of each mean value. Red dashed lines represent the minimum thresholds of ecosystem collapse (70%, range: 50-70%).

Future change (D2b)

We used expert estimates of *Grevillia australis* cover (shrub cover) and *Poa hemiata* cover (grass cover) documented in Camac et al. (2021) to determine present day (2017) and future (2050) values of shrub cover (Figure 23). Individual expert estimates were aggregated and presented as the mean and 95% confidence intervals. We assumed that the estimates for 2017 were representative of shrub and grass cover in 2000.

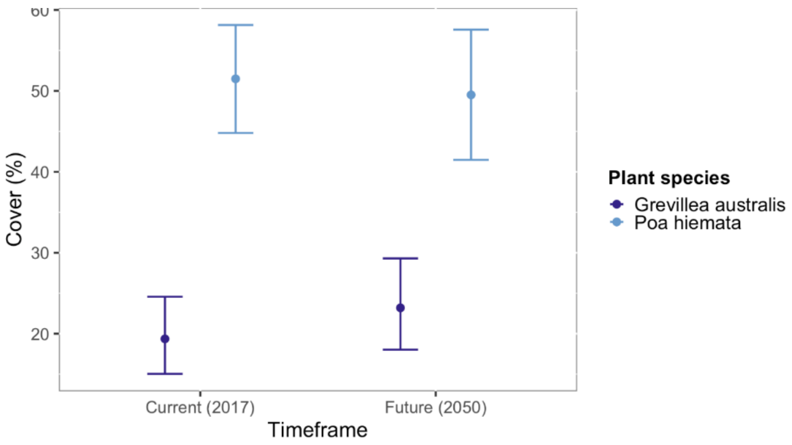


Figure 23. Cover of characteristic shrub *Grevillea australis* and grass *Poa hemiata* derived from expert estimates and used to assess *Alpine-subalpine Open Grassy Heathland and Herbfield* under Criterion D2b. Estimates of cover (%) for 2017 and 2050 were obtained from Camac et al. (2021). Solid points represent mean values of shrub cover. Vertical lines represent 95% confidence interval of each mean value.

Historic change (D3)

We inferred change since 1750 based on studies on shrub cover over the last century (see sub-criterion D1; Table 14, Figure 23) and perceptions of experts involved in the current project based on their personal experiences working in the ecosystem type.

Calculation of severity and extent

Recent change (D1)

Overall, 15.5% of *Alpine-subalpine Open Grassy Heathland and Herbfield* is distributed in Victoria. Field survey data were only available for a 6-year period (2004 -2010) in unburnt, previously grazed areas of the Bogong High Plains (Table 14). These data show that shrub cover is relatively low, although has increased over the six-year timeframe (from 5% to 11% cover). Aerial photography analysis suggests that 32% of the initial ecosystem extent on the Bogong High Plains, Victoria (including 3% without shrubs and 29% with shrubs) transitioned into *Alpine-subalpine Closed Heath* between 1936 and 1980 (McDougall 2003). However, the ecosystem type also increased in extent during this monitoring period: *Alpine-subalpine Closed Heath,* and other alpine ecosystems, had partially transitioned into *Alpine-subalpine Open Grassy Heathland and Herbfield* by 1980. The total area *Alpine-subalpine Open Grassy Heathland and Herbfield* did not substantially change during this monitoring period. In addition, regional spatial analysis of the mainland Australian Alps region suggests that shrub lines are encroaching into higher elevation grasslands, although this may be offset by upward treeline movement at lower elevations (Verrall et al. 2023).

Alpine-subalpine Open Grassy Heathland and Herbfield in Tasmania accounts for 9.45 % of the total ecosystem distribution. Data on shrub cover changes in Tasmania were only available from a two-year monitoring study (Table 14). Shrub cover did not change over the two-year period; however this evidence is insufficient to reasonably extrapolate across the entire assessment timeframe.

The NSW and ACT extent of *Alpine-subalpine Open Grassy Heathland and Herbfield* represents ~75 % of the total distribution. Averaging the values from Gungartan and Kosciuszko National Park (Table 14), shrub cover increased from 7.2 % ± 2.7 % (initial shrub cover) to 19.9 % ± 6.2 % (final shrub cover) between 1968 and 2001 (Table 14) (Scherrer & Pickering 2005). We assumed that estimated mean shrub cover in 2001 was a reasonable approximation of the present-day cover patterns across NSW and the ACT. Considering the collapse threshold of 70% shrub cover (range: 50-70%), the relative severity of shrub cover change is 20.22 % ((7.2 - 19.9)/(7.2 - 70) × 100), with a lower bound of 15.21% ((8.20 - 17.6) / (8.2070) × 100), and upper bound of 36.53% ((6.20 - 22.2) / (6.20 - 50) × 100).

We assumed the data (and calculated elative severity) from Scherrer & Pickering (2005) were representative entire ecosystem extent in NSW, ACT and Victorian extents (90.55 %). Together, this equates to a risk status of **Least Concern (Least Concern – Neat Threatened)** under sub-criterion D1.

Future change (D2b)

Past trends of climate-driven shrub cover change are expected to continue (Wahren et al. 2013; Camac et al. 2017). The increasing frequency of disturbance events (e.g., drought and fire) will increase the abundance of bare ground available for shrubs (and other flora) to colonise (Camac et al. 2015, 2017; Venn et al. 2017; Hickman et al. 2024) Evidence from *in-situ* experimental warming studies demonstrated that warmer temperatures of ~ 1 °C increased the growth rates of Australian alpine shrub seedings; faster spring and summer growth may improve long-term seeding survival during the harsher winter months (Camac et al. 2017). Fires are unlikely to increase to a frequency that is detrimental to the persistence of shrubs (see criterion C).

Experts estimated that the cover of the characteristic shrub species *Grevillea australis* may slightly increase between 2017 and 2050, while the cover of characteristic grass species *Poa hemiata* is likely to remain stable (Figure 23); the cover of shrubs in grasslands and heathlands will remain stable (± 2 %) (Camac et al. 2021). However, it is unlikely that shrub cover will increase by 2050 to meet the threshold for Vulnerable. Therefore, the ecosystem type is classified as **Least Concern** under sub-criterion D2b.

Historic change (D3)

Data prior to 1978 suggest that shrub cover has remained stable or slightly increased throughout the ecosystem extent. These increases in shrub cover have been driven by livestock and introduced species (McDougall 2003); conversely, declines in shrubs likely occurred between 1840 and 1960 as shrubs were targeted by pastoralists using fire to clear land for grazing (Zylstra 2006; Kirkpatrick & Bridle 2013). It was deemed highly unlikely that shrub cover would have increased sufficiently to meet the threshold of Vulnerable (> 50 % change). Therefore, the ecosystem type is considered **Least Concern** under sub-criterion D3.

Indicator: Exotic ungulate abundance

Relevance to ecosystem function

Native herbivores such as herbivorous rats and insects migrate during the growing season, providing an important food source for insectivorous birds and raptors (Green & Osborne 2012). However, exotic herbivores such as deer, pigs, horses and rabbits severely impact alpine ecosystems by grazing and trampling native vegetation, compacting soils, increasing bare ground, erosion and spreading invasive plant species (Hone 2002; Green & Osborne 2012; Claridge 2016; Environment and Communications Reference Committee 2023). Invasive herbivores also compete with native herbivores for food; there are few predators of rabbits in alpine regions (larger raptors may consume some rabbits) but there are no predators of horses, deer and pigs, and the presence of these fauna can substantially alter tropic food webs and nutrient cycling throughout the alpine region.

Prolonged periods of warmer temperatures and earlier snowmelt would drive earlier migration and prolong the residence times of both native and invasive herbivores. The increasing abundance, distribution and residence times of herbivores in *Alpine-subalpine Open Grassy Heathland and Herbfield* would reduce the growth and development of seed necessary to sustain the persistence of flora; collapse is likely if herbivory rates exceed capacity of flora to recover. Collapse may also occur if preferential herbivory drives declines in a few species: increasing shrub cover in Tasmania has been observed following the suppression of extents subject to rabbit grazing (Kirkpatrick & Bridle 2016). Grazing, coupled with the risks associated with large invasive herbivores (trampling, erosion, seed dispersal, soil compaction) would exacerbate the current threats to the ecosystem type (e.g., fire, drought) and lead to ecosystem collapse.

Data availability and quality

Current quantitative or qualitative evidence of distribution and abundance of invasive herbivores and native herbivores is limited. Exotic fauna are now the dominant herbivores in high elevation mainland extents (Hartley et al. 2022) although detailed information of herbivore extent and residence times is not available. If a minimum density (individuals per unit area, or herbivory rate per unit area) causing collapse can be determined, and if coupled with more comprehensive information on current and predicted future distributions and densities of native and invasive herbivores, then herbivory rates, and the abundance of native and / or invasive herbivores would be a useful indicator for future assessments.

Indicator: Grass and forb cover

Relevance to ecosystem function

Alpine-subalpine Open Grassy Heathland and Herbfield is characterised by the dominant cover of grasses, forbs and herbs. The loss of this non-shrubby vegetation, or dominance of invasive exotic flora species would represent ecosystem collapse.

Data availability and quality

There is limited information available on non-shrubby vegetation cover and evidence from field studies from the Victorian ecosystem distribution was insufficient to extrapolate across the entire ecosystem extent (Wahren et al. 2013; Camac et al. 2013, 2015). Processes and collapse pathways associated with the loss of grass and forb cover were considered to have already been captured by indicators of shrub cover, bare ground indicators (Criterion D) and indirectly through fire frequency (Criterion C). However, the cover of grasses and forbs following introduction of a novel threat (e.g., pathogen, novel invasive herbivore) may be useful to inform future assessments. The cover of invasive flora may also be useful to inform future assessments, particularly when coupled with disturbance events that create bare ground (for recruitment) and where vectors of seed dispersal (hikers, horses, vehicles, invasive fauna) are present.

Criterion E

Summary

No stochastic models are available and there are presently insufficient data to reliably inform model simulations. Therefore, the risk status is **Data Deficient** under criterion E.

The diversity of plant communities and lifeforms within *Alpine-subalpine Open Grassy Heathland and Herbfield* Photos: Susanna Venn.

Alpine-subalpine Closed Heath

Authors

Clare Vernon, Chloe Sato, Jessica Rowland, Emily Nicholson, Ella Plumanns Pouton, Kate Umbers

Reviewers

Jayne Balmer, James Camac

Contributors

James Camac, Richard Williams, Keith McDougall, John Morgan, Susanna Venn

Biome

T6 Polar-alpine

Functional group

T6.4 Temperate alpine grasslands and shrublands

IUCN status

Least Concern

Assessment Summary

Alpine-subalpine Closed Heath is defined by a dense canopy of dominant shrubs up to 2.0 m tall (≥ 70 % shrub coverage). It is restricted to the Australian mainland and occurs above the alpine treeline (> 1100 m). Predicted warmer and drier climate conditions are likely to promote expansion of *Alpine-subalpine Closed Heath* into neighbouring ecosystem types due to the spread and thickening of shrubs. However, plant pathogens such as *Phytophthora*, introduced ungulates, and changing fire regimes may also pose a threat to this ecosystem type. The risk status of this ecosystem type is assessed as **Near Threatened** due to recent changes in fire frequency (Table 15).

Table 15. Summary of the Red List of Ecosystems assessment of *Alpine-subalpine Closed Heath*. Category ranges in parentheses reflect uncertainty in assessment under the corresponding criteria.

Criteria	**A**	**B**	**C**	**D**	**E**	**Overall**
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	LC	NT	LC	DD	**NT**
Sub-criterion 2 *A, C, D: 50-year period including present & future* *B: AOO*	LC	LC	LC	LC		
Sub-criterion 3 *since ~1750* *B: number of locations*	LC	LC	LC	LC		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes ; E = quantitative probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria.

Ecosystem Description

Ecosystem Classification

In the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022a), *Alpine-subalpine Closed Heath* is a Level 6 sub-global ecosystem type, belonging to Ecosystem Functional Group *T6.4 Temperate alpine grasslands and shrublands* within the *T6. Polar/alpine (Cryogenic)* biome.

Under the National Vegetation Information System (NVIS) 6.0, the ecosystem type aligns with *Major Vegetation Group (MVG) 30 (Heathlands)* and *Major Vegetation Subgroup (MVS) 18 (Heathlands)*. In Victoria, the ecosystem type is referred to as "Mainland Heath" (McDougall 1982; Costin et al. 2000; Venn et al. 2017), "Subalpine Heath, "High Altitude Closed Heathland", "Bogong High Plains Closed Heathland", "Central Victoria Alps rocky open heathlands" (McDougall & Walsh 2007), and "Sub-alpine Shrubland", "Sub-alpine Dry Shrubland", "Alpine Rocky Outcrop Heathland", and "Alpine Dwarf Heathland" (Mackey et al. 2015).

In New South Wales (NSW), *Alpine-subalpine Closed Heath* is mapped as "Sub-alpine Dry Rocky Shrubland" and "Alpine Rocky Low Open Heathland" (Mackey et al. 2015) and referred to as "alpine heaths" (Keith 2004). Under the Murrumbidgee Vegetation Classification Scheme, *Alpine-subalpine Closed Heath* is inclusive of the following vegetation communities: *Bossiaea foliosa – Cassinia monticola – Kunzea muelleri – Hovea montana Heathland*, *Epacris* sp. – *Pentachondra pumila – Poa fawcettiae Heathland*, and *Prostanthera cuneata – Orites lancifolius – Nematolepis ovatifolia Heathland* (Armstrong et al. 2013). The ecosystem type shares characteristic features with and may also meet the classification of *Micromyrtus – Kunzea muelleri – Kunzea ericaites Dry Heathland* on skeletal ridges primarily of the Namadgi region (Armstrong et al. 2013).

Distinction from other ecosystem types

Alpine-subalpine Closed Heath is only found on mainland Australia and is characterised by a dense and dominant (≥ 70% cover) upright shrub canopy (0.5 - 2.0 m tall) with typically rocky substrates (Williams & Ashton 1988). Characteristic shrubs of this ecosystem type display fire-tolerant traits, including fire-stimulated germination (obligate seeding) and post-fire resprouting. This ecosystem often mosaics with other ecosystem types and often does not exhibit a defined ecosystem boundary. *Alpine-subalpine Closed Heath* intergrades with open heathland communities associated with *Alpine-subalpine Open Grassy Heathland and Herbfield*. This ecosystem type occurs on shallower slopes at higher elevations across the Australian mainland and Tasmania, and may be differentiated from *Alpine and subalpine closed heathland* by the presence of an open shrub cover (0 – 30%; shrubs are usually short [< 1.5 m] or prostrate in form), and the dominance of herbs and grasses (Williams & Ashton 1988; McDougall & Walsh 2007). *Alpine and subalpine Closed Heathland* also intergrades with *Fjaeldmark* ecosystems; the latter differentiated by a dominant cover of exposed rocks (> 50%) and windswept, prostrate shrubs, short grasses or herbs. *Alpine and sub-alpine Closed Heathland* only occurs on the Australian mainland; the analogue ecosystem in Tasmanian extents is the *Tasmanian Alpine Heathland*. This ecosystem is characterised by species endemic to Tasmania which often lack fire tolerant traits. The canopy is usually closed but where open, has an understorey comprising a mix of rushes, sedges, and forbs (Kirkpatrick 1983). *Subalpine Woodland and Forest* is often found adjacent to *Alpine-subalpine Closed Heathland* at lower elevational bands at or below the climate treeline. It is characterised by a taller canopy of small (< 12.0 m) snow gums (*Eucalyptus pauciflora)* which form the defining structural component of the ecosystem, and an understorey of grasses, herbs and shrubs (0.4 m - 4.0 m) (Venn et al. 2017).

Distribution

Alpine-subalpine Closed Heath occurs in alpine and treeless subalpine zones of mainland south-eastern Australia (Mackey et al. 2015). The ecosystem type occurs between –146.43° and –148.88° longitude and –35.58° and –37.45° latitude and occupies an altitudinal band of 1100 m to 2000 m above sea level. In total, *Alpine-subalpine Closed Heath* covers an area of 89.09 km^2 (Figure 24), including 37.07 km^2 in New South Wales, 5.90 km^2 in the Australian Capital Territory, and ~ 47.65 km^2 in Victoria. Spatial products used to develop the distribution maps of *Alpine-subalpine Closed Heath* included the most extensive and/or accurate mapping available at the time of assessment (May 2020; Appendix 1), as advised by government representatives involved in the assessment process.

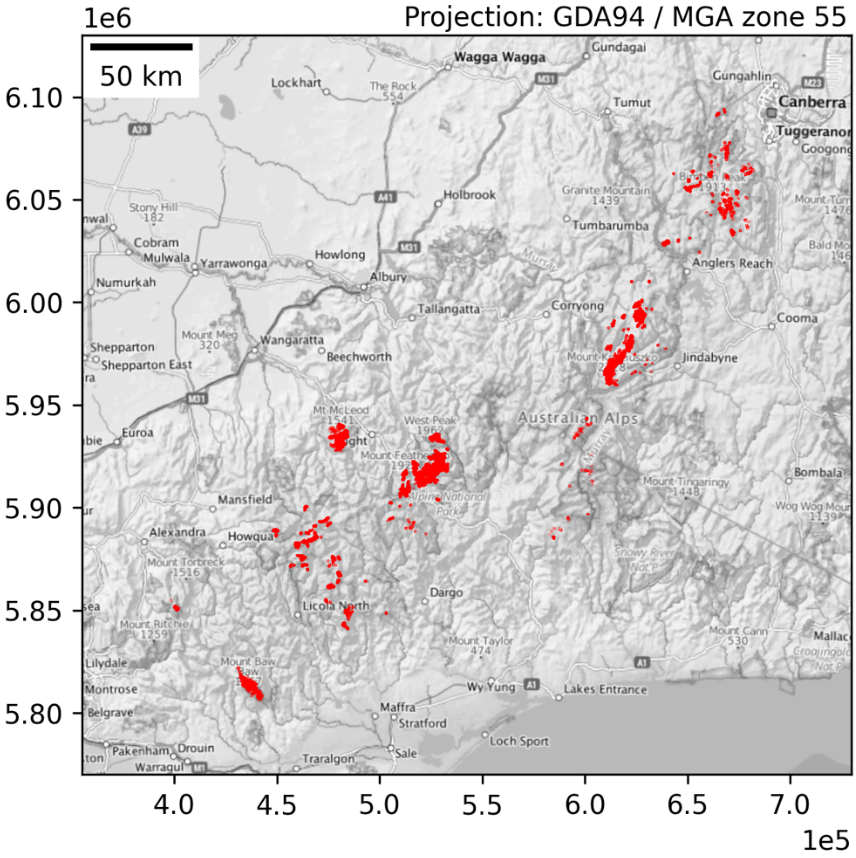


Figure 24. Extent of *Alpine-subalpine Closed Heath* (red) on mainland Australia.

Abiotic Environment

Alpine-subalpine Closed Heath is characterised by historically low annual average temperatures (< 8°C) (Venn et al. 2017) and high annual precipitation (average annual rainfall 606-2344 mm) (Stern et al. 2000). Precipitation typically occurs during winter and spring (June to September), and can occur as rainfall or snowfall (Venn et al. 2017). There is no distinct dry season at higher elevations (Stern et al. 2000). The spring snow melt provides an important source of groundwater recharge for *Alpine-subalpine Closed Heath* (McDougall et al. 2015).. Snowpack during winter often provides an insulating layer, preventing freezing of the soil, vegetation, roots, and soil fauna (Green & Osborne 1994). Severe winds and frosts are common (Williams et al. 2014; Venn et al. 2017).

Alpine-subalpine Closed Heath typically occurs on exposed rocky slopes with shallow, nutrient-poor soils (Mackey et al. 2015) (Figure 25). Bare ground cover is generally low in undisturbed patches (< 3%)(Camac et al. 2013; Williams et al. 2014). Fire, drought, and herbivory increase bare ground and it may take 10 years or longer for vegetation cover, bare ground cover and litter cover to return to pre-fire levels (Wahren et al. 2001).

Figure 25. *Alpine-subalpine Closed Heath* adjacent to *Alpine-subalpine Woodland and Forest* near Mt Jagungal, Kosciuszko National Park (NSW). Photo: Keith McDougall.

Characteristic Native Biota

Alpine-subalpine Closed Heath is characterised by a high cover (≥ 70 %) of evergreen sclerophyllous shrubs, 0.5-2.0 tall with a dense, upright canopy (Lamont & Keith 2014; Williams et al. 2014) (Figure 26). Small patches of grasses, forbs, herbs, and prostrate dwarf shrubs (0.1-0.5 m tall) are typical, although this represents < 10 % total cover (Camac et al. 2013). A few stunted *Eucalyptus pauciflora* and *E. niphophila* may occur in this ecosystem type. Species composition varies throughout the ecosystem extent (Armstrong et al. 2013), but is dominated by shrub species, including: *Orites lancifolia* (Proteaceae), *Grevillia australis* (Proteaceae), *Prosanthera cuneata* (Laminaceae), *Bossiaea foliosa* (Fabaceae) and *Phebalium squamulosum* (Rutaceae) (Camac et al. 2013). The understory vegetation is sparse due to limited light penetration and accumulation of a deep layer of leaf litter. The understorey is mainly comprised of dwarf upright or prostrate shrubs (such as *Olearia spp*., *Oxylobium ellipticum* and *Hovea montana*), forbs, sedges (e.g., *Carex hebes*) and non-tussock grasses (e.g., *Poa hothemensis*) (Armstrong et al. 2013; Lamont & Keith 2014).

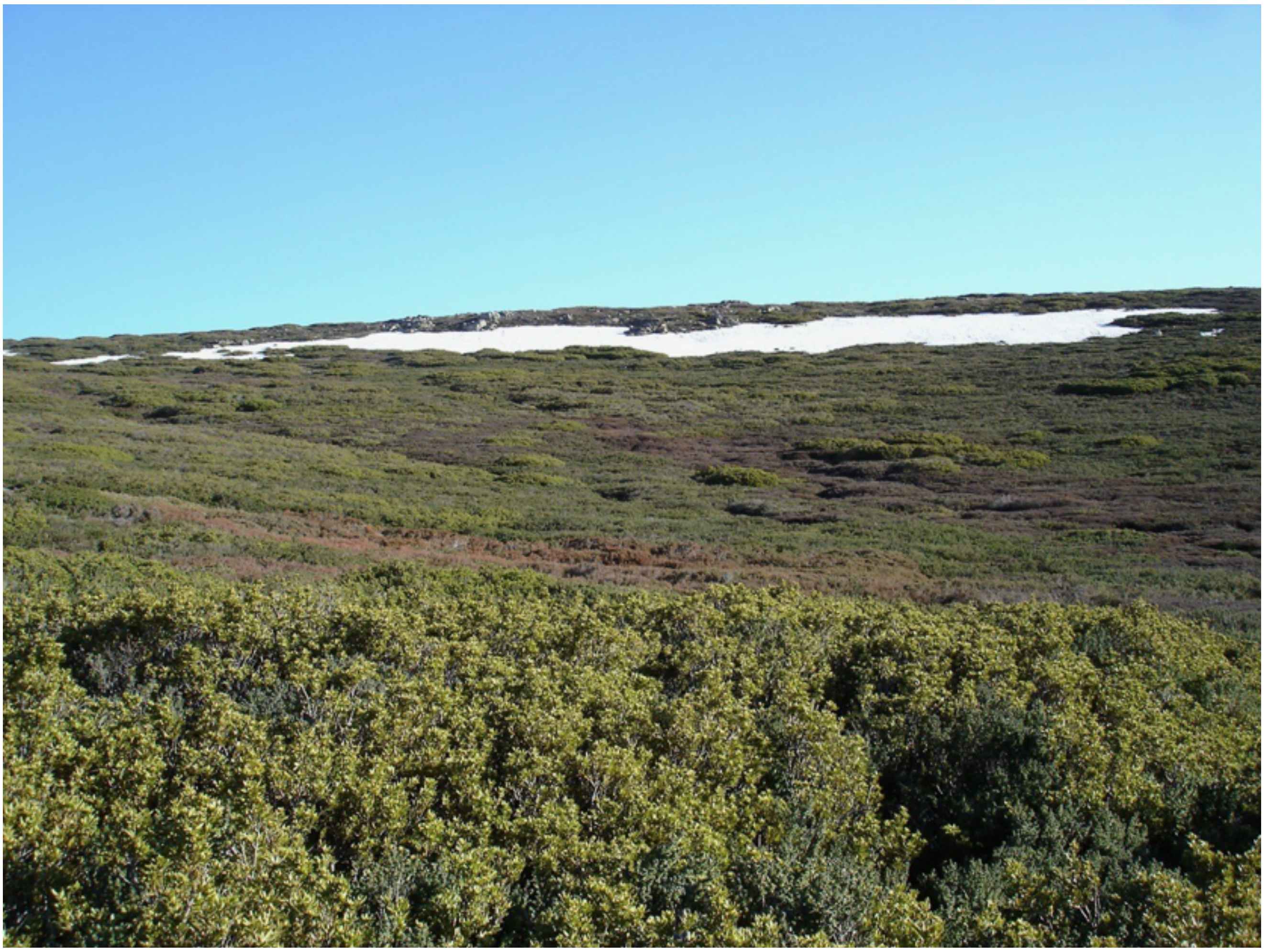


Figure 26. *Alpine-subalpine Closed Heath* showing dense shrub coverage. Photo: James Camac, 2021.

Alpine-subalpine Closed Heath supports a wide range of faunal species. Areas with higher structural complexity provide habitat for small mammals, such as the broad-toothed rat (*Masyacomys fuscus*), dusky antechinus (*Antechinus swainsonii*), brown antechinus (*Antechinus stuartii*), and bush rat (*Rattus fuscipes*) (Green & Osborne 1994; Green & Sanecki 2006). Birds are uncommon, but if present are typically seasonally migratory (Green & Osborne 1994; Green 2002), including scrub wrens (*Sericornis frontalis*) and Richard's pipits (*Anthus richardi*). Typical reptiles include the white-lipped snake (*Drysdalia coronoide*), mountain-log skink (*Pseudemoia entrecasteauxii*) and Guthega skink (*Liopholis Guthega*) (Green & Osborne 1994). The flowering of shrubs in during the warmer growing season ttracts pollinating moths, butterflies, bees, flies and grasshoppers (Green & Osborne 1994).

Interactions and Processes

Cold, snowy climate and seasonality

Alpine-subalpine Closed Heath occurs in exposed environments with harsh climatic conditions and steep environmental gradients as a product of the sloping, mountainous topography. Distinct seasons occur within the alpine region: productivity increases during the short growing seasons (spring and summer months) where sufficient thermal and sunlight resources enabled growth, flowering and reproduction. Winter productivity is limited, with prolonged snow cover constraining the thermal and light energy available for growth. Local climatic conditions are moderated by topography, aspect and slope and have a greater influence on flora and ecosystem distribution than regional, macroenvironmental conditions (Lim et al. 2017). Snow accumulation during winters is supported on the closed, dense shrub canopy insulating both shrub and understorey vegetation from frost, wind and cold, desiccating conditions (Sturm et al. 2001; Myers-Smith et al. 2011). Most fauna migrate to lower, warmer elevations in winter or seek shelter *in-situ* under the snowpack (e.g., spotted mountain grasshopper (*Monistria concinnai*), the mountain heathland dragon (*Tympanocryptis* diemensis) and the broad-toothed rat (*Mastacomys fuscus*) (Green & Osborne 1994). Freeze-thaw cycles before the onset of winter (and accumulation of sufficiently deep, insulating snowpack) can drive the formation of needle ice in more open extents of *Alpine-subalpine Closed Heath*, uplifting soils and causing frost heave. This process exposes shrub roots to desiccating cold conditions, damaging root structures and causing the shrub to senescence if severe. Most extents of the ecosystem occur on steep, exposed locations (where needle ice is less likely to form). Further, the accumulation of a deep litter layer under the shrub canopy insulates the soils and buffers against needle ice formation (Green & Osborne 1994; Venn et al. 2021). As germinant flora relies on frost hardening to survive the cold winter conditions, the mortality of establishing shrubs (which often occur in bare ground patches, or patches without deep litter layers) may increase until this physiological tolerance is developed (Venn & Green 2018). Snowpack also influences surface vegetation, soil, and atmosphere energy exchanges by increasing the surface albedo (Green & Osborne 1994; Green & Pickering 2009b; Treby et al. 2024); snowmelt occurs sooner in extents withe taller, denser shrubs are present. Warming temperatures during spring and summer determine the onset of snowmelt, providing an important release of nutrients and groundwater recharge to supporting most flora and fauna (Green & Osborne 1994; Morgan & Venn 2017). These seasonal windows are an important food source for native migratory species during the growing season.

Recruitment

Alpine-subalpine Closed Heath is characterised by cyclical succession patterns (Williams & Ashton 1987). The lifespan of shrub species varies; senescence in some species may occur as soon as 20 years, while others are thought to persist for over 100 years (e.g., *Orites lancifolia*). Recruitment is reliant on disturbance events to create bare ground, stimulate germination, and provide an opportunity for recruitment (Williams & Ashton 1988; Green & Osborne 1994; Venn et al. 2017, 2021). *Alpinesubalpine Closed Heath* exhibits cyclical succession (Wahren 1997): bare ground colonisation by fast-growing *sedges (e.g., Carex),* grasses (e.g., *Poa hothemensis* at Bogong High Plains, Victoria and obligate seeder shrubs. As light penetration through the canopy and space between shrub canopies declines with shrub growth, and the establishment of new vegetation shifts from *Poa* to shrubs. Mature shrubs act as “nurse shrubs”, protecting smaller shrubs from freezing winter conditions and providing shade during summer. Eventually, a dense, continuous upper canopy and lower litter-filled understorey is formed, with few grass, sedge and forb individuals present (Williams & Ashton 1987).

Fire

Alpine-subalpine Closed Heath is the most flammable treeless ecosystem type in the Australian alpine region (Williams et al. 2006; Fraser et al. 2016). Shrubs are characterised by oil-rich, highly flammable canopies (e.g., *Bossiaea foliosa*, *Orites lancifolia and Prostanthera cuneata*) and make dense leaf litter. Long time periods between fires allows dense, aerated fuel loads to accumulate (Fraser et al. 2016). Although infrequent (once a century), fires are often severe (Williams et al. 2014) and usually ignited by lightning (Williams et al. 2008). These fires typically burn > 80% of biomass (Fraser et al. 2016; Murphy et al. 2019; Foulkes et al. 2021), reducing vegetation to the mineral soil layer (Williams et al. 2006; Camac et al. 2013) (Figure 27). It may take 10 years or longer for bare ground and vegetation structure to return to pre-fire levels (Camac et al. 2013; Verrall & Pickering 2020). High temperatures during summer and/or repeat fires may increase the mortality of establishing or resprouting shrubs, as bare ground exposes roots to extreme heat (Wahren et al. 2013).

Figure 28. Shrubs following fire where canopy cover has been burnt to the mineral layer (top). Shrubs are more flammable than grasses (below). Photos from (Camac et al. 2021).

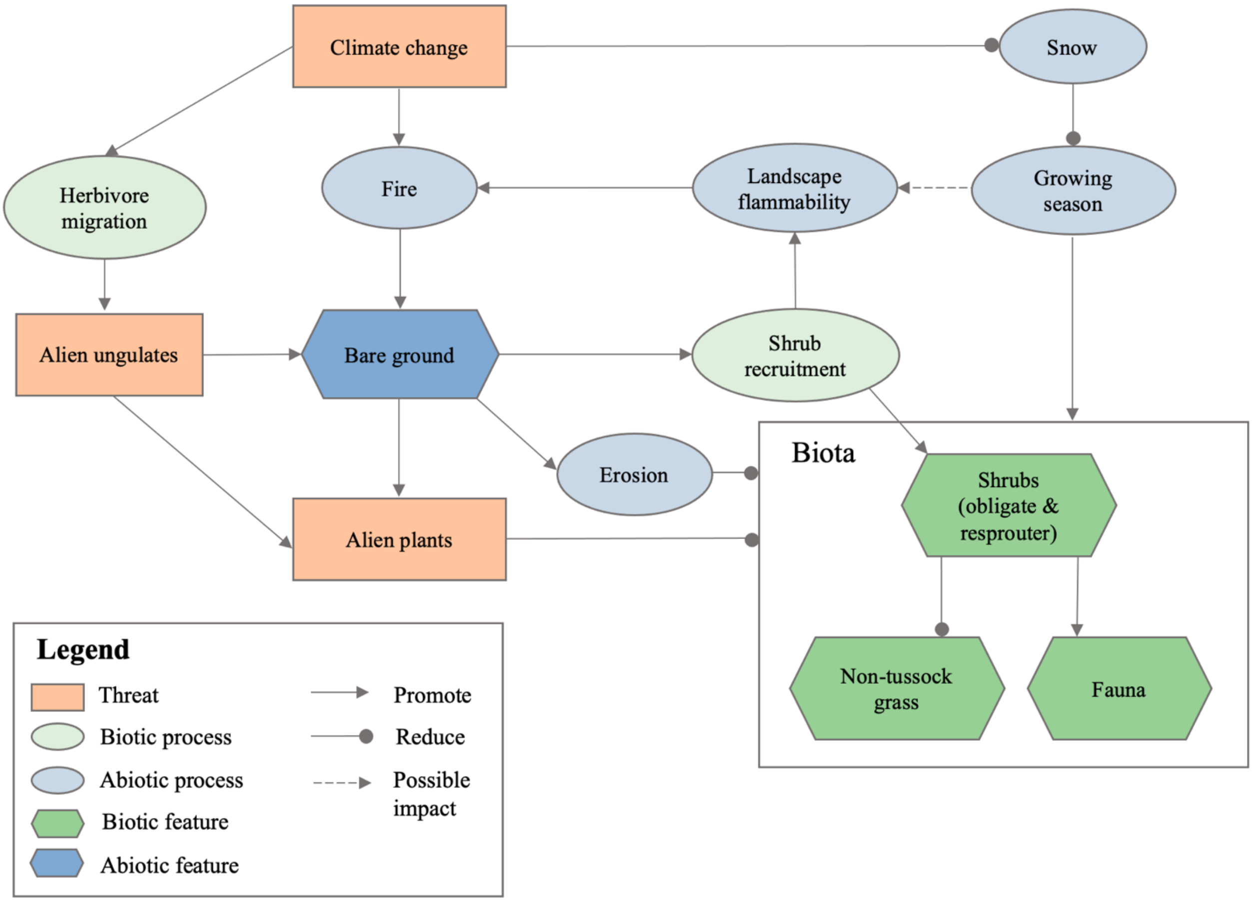


Figure 29. Conceptual model for *Alpine-subalpine Closed Heath*.

Major Threats

Climate change

Climate change is the primary threat to Australian alpine and subalpine ecosystems (Figure 29) (Camac et al. 2021). Declines in precipitation, increasing temperatures, and declining snowpack duration and depth have been recorded since the 1950s (Sánchez-Bayo & Green 2013). Prolonged drought and higher ambient temperatures have been associated with an increase shrub mortality (Leigh et al. 1987; Griffin & Hoffmann 2012; Williams et al. 2014).

The impact of increasing temperatures on *Alpine-subalpine Closed Heath* is uncertain. Temperature increases coupled with reduced precipitation, and soil moisture (Pickering 2007; Hennessy et al. 2008; Ji et al. 2022) will drive declines in the depth and persistence of the winter snowpack (Hennessy et al. 2008; Sánchez-Bayo & Green 2013). The loss of insulating snowpack will expose soil, plants, and fauna that remain in-situ during winter to freezing conditions (Williams et al. 2014). Conversely, the once range-limiting thermal constraints coupled with increasing thermal energy will support shrub growth, in-situ thickening and encroachment (Camac et al. 2017; Verrall 2023). The shrubs characteristic of *Alpine-subalpine Closed Heath* is tolerant of fire, drought and frost; climate change will also drive increases in the frequency of these disturbance events. As colonisation in alpine ecosystems relies on the presence of bare ground, climate change will drive likely drive both recruitment opportunities, accelerate shrub growth and height, and shorten the time to reproductive maturity (Elmendorf et al. 2012; Camac et al. 2017). Shrub encroachment has already been recorded throughout the Australian alps (Camac et al. 2017; Verrall 2023) driving declines in S*nowpatch Herbfield* (Williams et al. 2015) and in-situ thickening of *Alpine-subalpine Open Grassy Heathland and Herbfield* (Wahren et al. 2013).Long-term shrub expansion may also drive declines in *Fjaeldmark* ecosystems if the cover of shrubs increases such that the defining cover of rocks (> 50 %) is lost (Elmendorf et al. 2012).

Warmer conditions may also enable in-situ thickening of sparse *Eucalyptus*; shrub encroachment upwards is likely being offset by treeline advance throughout the alpine region (Verrall 2023). Warmer conditions may also induce earlier shrub flowering (Green 2010; Venn et al. 2017), leading to a mismatch between flower emergence and pollinator migration. A warmer, more favourable climate may also facilitate upward migration of fauna (Green 2010), including exotic ungulates (Claridge 2016), exacerbating declines due to herbivory, grazing (Green 2010) and dispersal of both exotic flora and disease (McDougall & Walsh 2007; Rigg et al. 2018). Concurrent threats such as drought, frost and fire may also increase shrub mortality and thus buffer constrain climate-driven shrub expansion (Leigh et al. 1987; Williams 1990; Enright et al. 2015). Shrubs may also require longer intervals to recover due to the increase physiological stress of repeat fire events (Enright et al. 2015). This may lead to lower species richness and a compositional shift towards fewer, more fire-tolerant shrub species that exhibit tolerance to the new environmental conditions at a given site.

Novel fire regimes

Fire intervals shortened substantially following colonisation, from 90-120 years, to 3.5-7 years (Zylstra 2006) as pastoralists used fire to clear heathlands for pastoral land practices (Williams et al. 2008). More recently, climate change – via warmer temperatures and reduced precipitation - has driven significant changes in the frequency of fire weather days and fire widespread fire events (Zylstra 2006; Pickering 2007; Clarke et al. 2011; Williams et al. 2014). Short fire intervals (< 10 years) may not allow sufficient time for obligate seeding shrubs to reach reproductive maturity and accumulate a soil or canopy-stored seedbank. Without a viable seedbank, obligate seeding shrubs are unable to persist post-fire (Enright et al. 2015) and local population extinction is likely Short fire intervals may also increase the regenerative stress on resprouting shrubs, . The physiological stress of resprouting mature and germinant shrub (and thus mortality) may be exacerbated by repeat fires, and functional collapse may occur due to resprouter failure (Enright et al. 2015; McDougall et al. 2015; Camac et al. 2017). Consequently, very short fire intervals (< 10 years) may transition *Alpine-subalpine Closed Heath* to *Alpine-subalpine Open Grassy Heathland and Herbfield*, which is characterised by the dominant biota of herb, forb and grass species with short timeframes to reproductive maturity (Williams et al. 2014). As bare ground is the primary recruitment mechanism for flora, short fire interval may facilitate colonisation by increasing the bare ground, and thus, providing recruitment & establishment opportunities for both exotic and endemic species from different ecosystems (Williams et al. 2014; Van Klinken & Friedel 2017; McDougall et al. 2018).

Invasive plant species

Invasive flora, such as *Acetosella vulgaris* (McDougall 2007) outcompete native vegetation (IPBES 2019) and persist in the soil seedbank for many decades (McDougall et al. 2018). Colonisation is reliant on a mechanism to introduce the seeds of invasive flora into the landscape and disturbance event to provide opportunity for recruitment (Hickman et al. 2024). Roads, ski villages and resorts built for tourism increase the opportunity for introduction and dispersal of invasive species by hikers or on vehicles (Williams et al. 2014; Van Klinken & Friedel 2017; McDougall et al. 2018; Pickering & Michael 2022). Fauna also disperse invasive flora throughout the landscape: increased movement as alpine regions coupled with more frequent fire and drought disturbances is likely to increase seed dispersal and recruitment opportunities.

Formation of a dominant canopy of *Eucalyptus pauciflora* following disturbance would represent a transition to *Subalpine Woodland and Forest* (Naccarella et al. 2020).

Large alien herbivores, browsing pressure

Livestock and horses are unlikely to enter intact *Alpine-subalpine Closed Heath* due to the dense canopy cover (Lamont & Keith 2014). Rather, grazing by introduced fauna is a significant threat to *Alpine-subalpine Closed Heath* (Williams et al. 2014), particularly deer (*Dama dama, Cervus elaphus, C. timoensis, C. unicolor, and Axis porcinus*; (Claridge 2016) and feral pigs (*Sus scrofa*) (Green & Osborne 1994; Nimmo & Miller 2007). Historically, livestock such as cattle, sheep and horses also posed a threat to flora due to trampling and grazing, and the indirect use of fire for pastoral activities which have left lasting legacies on the composition of alpine flora (Zylstra 2006; Moss 2024). Large invasive fauna cause significant structural damage via trampling and wallowing, opening the shrub canopy, increasing bare ground cover, promoting soil erosion, and spreading introduced plants (Williams et al. 2014). Warmer temperatures are likely to increase the residence times of both invasive fauna (e.g., deer and pigs) and native fauna in the ecosystem type, increasing the duration in which of wallowing, trampling and grazing can occur (Rivrud et al. 2019). Areas exposed to grazing following a fire event are likely to experience the largest declines in ecosystem condition.

Plant pathogens

Pathogens such as *Phytophthora cambivora*, *Armillaria spp.* and *Phytophthora cinnamomi* represent a threat to shrubs (McDougall et al. 2015; Green 2016; Rigg et al. 2018). These diseases cause root rot and wood decay, withdieback in shrub and woodland ecosystems already observed at lower elevations (Barrett & Yates 2015). *Phytophthora* has already exhibited cold adaptation, and instances have been recorded in regions where *Alpine-subalpine Closed Heath* occurs (e.g., Mt Kosciuszko, NSW at~ 1600 m) (Rigg et al. 2018; Khaliq et al. 2019). Warming temperatures and increased activity by humans and fauna (exotic, native) alike will increase the spread of these diseases throughout the alpine region in the future. change and vectors (e.g., hiking trails, roads) is likely.

IUCN Stress Classification

1.2 – Ecosystem degradation
1.3 – Indirect ecosystem effects

IUCN Threats Classification

1.3 – Tourism and recreation areas
2.3.1 – Nomadic livestock grazing
7.1.1 – Increase in fire frequency / intensity
8.1 – Invasive non-native species
11.1 – Habitat shifting and alteration
11.2 – Droughts
11.3 – Temperature extremes

Ecosystem Collapse

Ecosystem collapse in *Alpine-subalpine Closed Heath* can occur in three key ways.

- The ecosystem type transitions to *Alpine-subalpine Open Grassy Heathland and Herbfield* if shrub coverage declines to < 70% without signs of recovery within the expected timeframe (e.g., lack of shrub resprouting and/or germination ten years after a disturbance), and instead bare ground is infilled by fast-establishing grass, forb, and herb species; recovery from collapse would be dependent on recolonisation or expansion on recolonisation of shrubs from adjacent, intact patches.
- *Alpine-subalpine Closed Heath* may collapse into a novel depauperate ecosystem type after repeated short-interval fire (as observed at Bald Hill, Bogong High Plains, Victoria). Excessive soil erosion after fire would destroys the soil seed bank and prevent formation of viable germination beds for recovery (Enright et al. 2015).
- *Alpine-subalpine Closed Heath* may collapse and transition into *Sub-alpine Woodland and Forest Eucalyptus* establish: shrubs would become a closed heath understorey. Evidence of treeline encroachment (driven by warming climates) suggests that this collapse pathway is possible (Verrall 2023) and most likely to occur after a disturbance event (McDougall et al. 2018; Naccarella et al. 2020). However, Eucalyptus requires longer fire intervals to reach reproductive maturity (> 30 %), longer than the interval of shrubs and thus any encroachment may be offset by predicted increases in fire, drought and / or pathogens that would kill the establishing tree These pathways is considered least likely to cause collapse in the near future.

In this assessment, collapse in *Alpine-subalpine Closed Heath* occurs when:

1. *Area*: The mapped distribution declines to zero (100% loss) (Criteria A and B), or;
2. *Fire Frequency:* Fires increase to a frequency of ≥ 1 in every 10 years (range: 10-20 years), where shrub cover is < 50% 10 years after a disturbance (Criterion C), or;
3. *Bare Ground:* Cover of bare ground increases to ≥ 10% five years after a disturbance, where shrub cover is < 50% 10 years after a disturbance (Criterion C), and/or;
4. *Shrub cover:* Cover of shrubs increases to < 50% 10 years (10-15 years) following a disturbance (with no sign of recovery) (Criterion D).
5. *Tree cover:* trees are no longer stunted (> 4 m tall) and become the defining canopy in the landscape.

Risk Assessment

Criterion A: Reduction in distribution

Summary

The risk status of *Alpine-subalpine Closed Heath* is assessed as **Least Concern** under sub-criteria A1 and A2, and **Least Concern (Least Concern-Near Threatened)** under sub-criterion A3.

Methods

Recent change (A1)

Present day spatial data were sourced from the *ACT Vegetation Map 2018* (1:10000, 100 m with 83% accuracy) (Baines et al. 2013), *Forest Ecosystems: Vegetation of the Southern Forests VIS ID 3858* (1:25000, 250 m to 1:100000, 1 km archived at 2005, with accuracy considered reasonable to good as assessed by Gellie 2005), and *Native Vegetation – Modelled Extent 2005* (1:25000, 250 m, archived at 2005, with accuracy considered good although recommended scale at 1:100000) (Newell et al. 2006). A summary of spatial data, resolution, accuracy and process used to construct current extent maps is available in main methods.

Future change (A2)

Future change in area was inferred from patterns in past observed responses and experimental data showing responses to disturbances (fire and temperature change) (Camac et al. 2013, 2017). We also used 22 expert estimates of the current (2017) and likely future (2050) cover of key shrub species, including *Orites lancifolia, Prostanthera cuneata*, and *Grevillea australis*. Expert estimates from Camac et al. (2021) were aggregated and presented as mean shrub cover with associated 95% confidence intervals.

Historical change (A3)

Modelled historical distribution maps were available for NSW (*Forest Ecosystems: Native Vegetation of the Southern Forests: South-east Highlands, Australian Alps, South-west Slopes, and SE Corner bioregions. Pre 1750. VIS ID 3859*) and Victoria (*Native Vegetation – Modelled 1750 Ecological Vegetation*). Modelled past distribution for ACT were unavailable so excluded. To assess criterion A3, the present-day ecosystem distribution was compared to the modelled 1750 ecosystem distribution, and the percentage change used to determine risk outcome. Estimates of distribution change since 1750 also incorporated existing scientific knowledge of shrub cover change over the last century and expert knowledge.

Assessment outcome

Recent change (A1)

Historical livestock grazing and land clearing (c. 1850-1950) of *Alpine-subalpine Closed Heath* across the Bogong High Plains, Victoria, ceased in 2003 (McDougall & Walsh 2007). Based on observed changes in vegetation in the adjacent ecosystem type *Alpine-subalpine Open Grassy Heathland and Herbfield*, shrub and non-shrub vegetation cover likely increased following the end of grazing (c. 1960 in New South Wales and the Australia Capital Territory, 2003 in Victoria) (Wahren et al. 1994; Scherrer & Pickering 2005; Williams et al. 2006). However, shrub cover in grazed areas is likely lower than ungrazed, intact ecosystem extents (Wahren et al. 1994; Scherrer & Pickering 2005; Williams et al. 2006). Aerial photography suggested that the area of *Alpine-subalpine Closed Heath* increased between 1939 and 1980 (McDougall 2003). However, a small extent of *Alpine-subalpine Closed Heath* continues to be cleared for ski runs at Falls Creek, Victoria and Mt Hotham, Victoria (McDougall & Walsh 2007). The total 'skiable' area of the Australian alpine region is < 3 km; less than 1 % of the present-day distribution of the ecosystem type, thus localised ski run clearing is unlikely to lead to ecosystem-wide collapse. Overall, the ecosystem type is classified as **Least Concern** under sub-criterion A1.

Future change (A2)

The distribution of *Alpine-subalpine Closed Heath* is predicted to increase over the next 50 years. Warmer, drier future climates are likely to increase shrub growth, shrub density and the elevational range in which germinating shrubs can establish (Camac et al. 2017, 2021) although this upward encroachment (i.e. increasing distribution at higher elevations) may be offset by tree line advance (i.e. decreasing distribution at lower elevations) (Verrall 2023). Increasing disturbance frequency and severity, such as fire and drought, may drive mortality of groundwater-dependent flora and dry-grassland species such as *Poa spp.* (Griffin & Hoffmann 2012; Williams et al. 2014). This senescence, coupled with increasing bare ground will provide opportunities for shrub establishment in previously range-limited regions (Camac et al. 2017, 2021). Expert estimates predict an increase in the cover of shrubs, and spatial distribution of heathland ecosystems, inclusive of *Alpine-subalpine Closed Heath* (2017 and 2050) (Camac et al. 2021). Overall, the ecosystem type is stable and likely to increase in size and is thus classified as **Least Concern** under sub-criterion A2.

Historical change (A3)

There was no evidence of decline based on differences between current and modelled 1750 distribution in NSW and Victoria. Aerial photography analysis of the Bogong High Plains, Victoria shows that the area of *Alpine-subalpine Closed Heath* increased between 1939 and 1980 (McDougall 2003). This likely represents a conservative estimate of change in ecosystem distribution; the influence of livestock grazing in Victorian was more extensive and grazing ceased later (2003) than in the NSW and ACT ecosystem distributions (c. 1960s) (Scherrer & Pickering 2005; Williams et al. 2006).

Observed data suggest that the ecosystem type has likely experienced minimal net change in distribution since 1750. Overall, the ecosystem type is classified as **Least Concern** under sub-criterion A3.

Criterion B: Restricted distribution

Summary

Alpine-subalpine Closed Heath is assessed as **Least Concern** under sub-criteria B1, B2, and B3.

Methods

The Extent of Occurrence (EOO; B1) and Area of Occupancy (AOO; B2) of *Alpine-subalpine Closed Heath* were determined using a combination of existing map products from the Australian Capital Territory (ACT Government 2018), New South Wales (Gellie 2005; DELWP 2021), and Victoria (DELWP 2021). The data for subalpine treeless Victoria was augmented by experts with high resolution aerial photography from 2011 to better reflect on-ground ecosystem boundaries.

The number of threat-defined locations was based on fire, as this is the most important known stochastic threat to *Alpine-subalpine Closed Heath*.

Assessment outcome

Sub-criterion B1

The Extent of Occurrence (EOO) for *Alpine-subalpine Closed Heath* is 30,073.86 km^2 (Figure 30), meeting the threshold for Vulnerable. However, due to the lack of substantial ongoing declines, the risk status is **Least Concern** under sub-criterion B1.

Sub-criterion B2

Alpine-subalpine Closed Heath occupies 96 10 × 10 km grid cells (AOO; Figure 30), resulting in a risk status of **Least Concern** under sub-criterion B2.

Sub-criterion B3

The major stochastic threat for *Alpine-subalpine Closed Heath* is fire at very high (short) frequency intervals (< 10 years). Between 1970 and 2020, fires occurred in 22 fire seasons (from July to June), affecting a total of 77.87 km^2 (87.4% of the distribution); in addition, modelled estimates from Criterion C3 indicate fire burnt 0.34 km^2 between 1939 and 1960). Most fires have only burnt a small portion of the ecosystem type, although several large fires burnt extensive areas in 1938/1939, 2002/2003, 2006/2007, and 2019/2020; fires in 1938/1939 and 2002/2003 burnt multiple locations (2-10 distinct geographical locations) (Pickering 2007). It is unlikely that such large-scale fires would recur in intervals required to cause collapse (< 10 years) due to the patchy, dispersed nature of the ecosystem type and the high degree of resilience that characteristic, fire-adapted shrub vegetation exhibit (Camac et al. 2017, 2021). Therefore, the ecosystem type is considered **Least Concern** under sub-criterion B3.

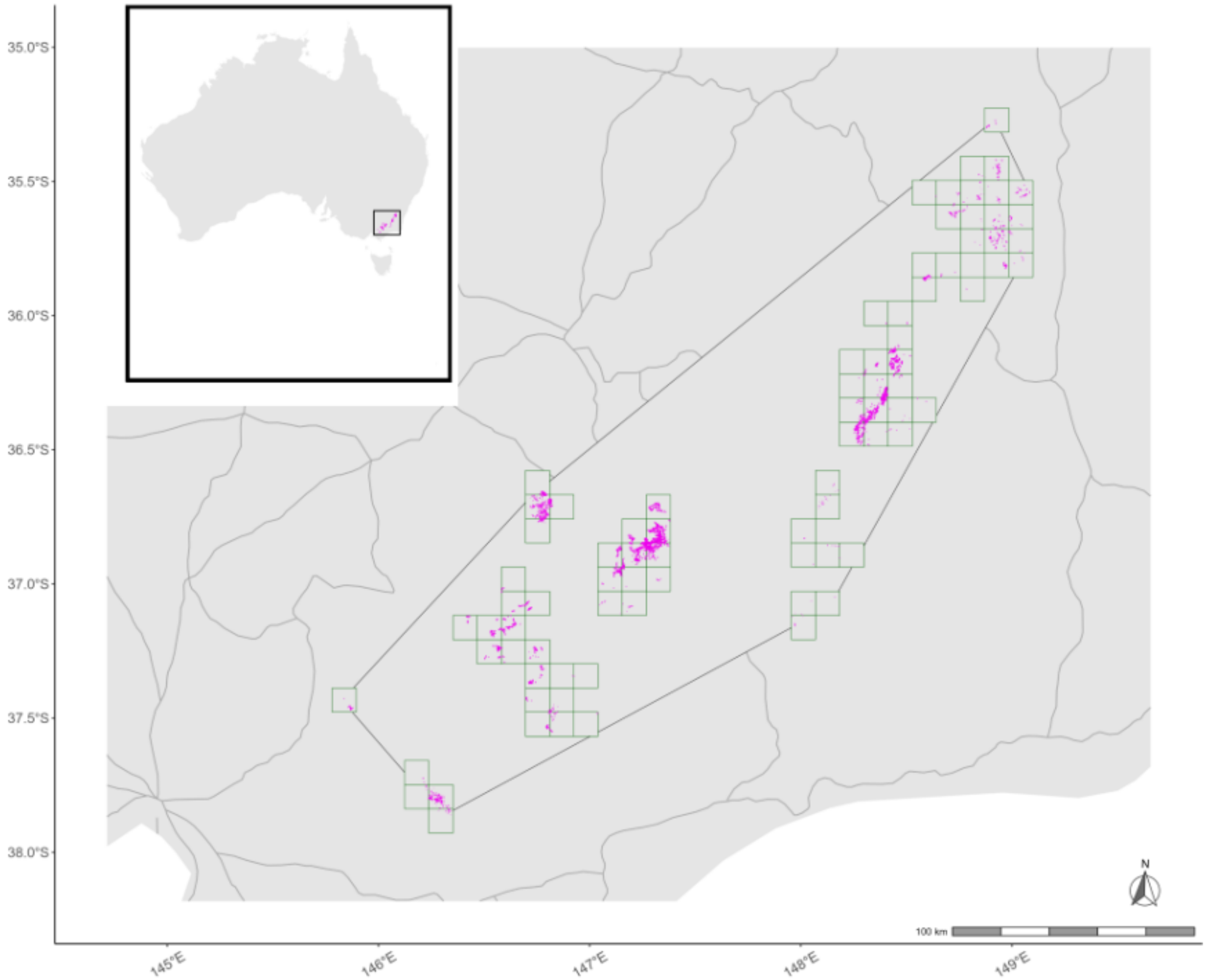


Figure 30. Map of *Alpine-subalpine Closed Heath* (magenta polygons) across Australia, showing Extent of Occurrence (black polygon) and Area of Occupancy where the 1% rule was not applied (green squares).

Criterion C: Environmental degradation

Summary

When assessed using fire frequency and bare ground, *Alpine-subalpine Closed Heath* meets the definition of **Least Concern** under all sub-criterions.

Identification of abiotic indicators

We considered two indicators to assess risk of collapse by environmental degradation:

- *Fire Frequency:* a measure of the frequency (years) in which the same area burns.
- *Bare ground*: a measure of groundcover lacking vegetation, but including leaf litter, rocks, and bare soil.

Indicator: fire frequency

Relevance to ecosystem function

Fire is a characteristic process in *Alpine-subalpine Closed Heath* (Williams & Ashton 1988) although fire events prior to 1750 were less frequent (interval pre-1750: 90 – 121 years) (Zylstra 2006). Repeated, short-interval fires represent a threat to characteristic shrub vegetation. Fire frequencies greater than the timeframe for obligate seeder and resprouter shrub species to reach reproductive maturity may lead to the local species extinction (Enright et al. 2015). Repeated short interval fires also place sustained physiological stress on resprouting shrubs and lead to resprouter failure (Enright et al. 2015; McDougall et al. 2015).

Data availability and quality

Recent change (C1) and Historical change (C3)

Historical records of bushfires and planned burns are available as shapefiles and detailed in the main methods.

Future (C2a)

We used FROST to create 100 replicate simulations of the 20-year period from 2060 to 2079 (Mccoll-Gausden et al. 2022). For each simulation, we extracted a spatial raster that displayed the number of fires burnt within the 20-year period, for each 180 m^2 cell. For each climate scenario, we calculated the percentage of the ecosystem type (Extent, based on the number of cells) that burnt at a frequency that exceeded the collapse threshold (assuming Relative Severity [RS] = 100%).

Selection of collapse threshold

Obligate seeding shrubs take at least 5 years to reach reproductive maturity, and repeated fire intervals of < 10 years likely to exhaust the soil seedbank and drive local population extinction (Camac et al. 2013). Therefore, ≥ 2 fires within 10 years (range: 10 – 20 years) across the whole ecosystem distribution would indicate ecosystem collapse, equating to a spatially weighted annual probability of burning of 0.20.

Selection of initial and present/future values

Recent change (C1)

To define the initial and current fire frequency, we analysed the fire history datasets from the Victorian and NSW state governments (NSW Goverment 2022; Victorian Government 2022b). We overlaid the fire history layers and extracted areas where the ecosystem type had burn. To generate a time-series, we calculated the number of times an area burnt between 1970 - 1979 (initial value), 2010 - 2020 (current value), and in 10-year intervals between these timeframes.

Future change (C2a)

We used FROST to create 100 replicate simulations of a 20-year period from 2060 to 2079. For each simulation, we extracted a spatial raster that displayed the number of fires burnt within the 20-year period, for each 180 m^2 cell. For each climate scenario, we calculated the percentage of the ecosystem type (based on the number of cells) that burnt at a frequency that exceeded the collapse threshold (i.e., where relative severity is 100%). We assumed that extents where the frequency exceeded 2 fires in 20 years were representative of the collapse threshold exceeded.

Historical change (C3)

We used the same approach as used in sub-criterion C1 to calculate initial and current values. We also considered evidence from published literature considering the location and extent of fire events, historical use of fire for land clearing and pasture.

Calculation of severity and extent

Recent change (C1)

Fires have burnt *Alpine-subalpine Closed Heath* in 22 fire seasons between 1970 and 2020, affecting a total of ~ 77.87 km^2, or ~ 87.40 % of the whole ecosystem extent. Overall, 76.44 % has burnt 1-2 times within the past 50 years, with only 3.18 % burnt > 3 times in the last 50 years (Table 16). The largest fires during the assessment timeframe occurred in 2003 and 2020, burning significant portions of the ecosystem in Victoria and Kosicuszko National Park, NSW. Even if the same sites burnt in 2003 were again burnt in 2020, this would represent a 16-year fire interval. The ecosystem has not experienced widespread fires at intervals less than the collapse threshold and is considered **Least Concern** under sub-criterion C1.

Table 16. The percentage (%) of the *Alpine-subalpine Closed Heath* area (based on the number of cells) that have burnt over the threshold from 1960 - 1980, 1980 - 2000 and 2000 – 2020 and the actual respective fire frequencies.

Period of 20-year interval	**Percentage of ecosystem burnt over threshold (2/20 years)**	**Number of fires**	**Percentage of ecosystem burnt**
1960-1980	3.04%	1	11.42
		2	3.03
		3	0.02
1980-2000	0.33%	1	9.40
		2	0.33
2000-2020	22.59%	1	64.44
		2	20.33
		3	2.25
		4	0.002

Future change (C2a)

Most of the ecosystem type did not burn at a frequency at or above the collapse threshold in any of the 100 simulations (Table 17). The mean predicted fire frequency was less than the collapse threshold and the highest predicted fire frequency was in the most northern part of the extent (Figure 31). Under all four climate scenarios, the area of ecosystem extent burnt at intervals at or exceeding the collapse threshold ranged from 0.04% - 3.99% of ecosystem extent, less than the threshold of Near Threatened (25 % Extent, assuming RS = 100 %). Regional projections of south-eastern fire frequency suggest there will be limited changes in the fire weather by 2050, although the duration and magnitude of fire-prone conditions is likely to increase by 2100 (Clarke et al. 2011) as the magnitude and duration of weather conditions conducive to fire increase throughout the alpine region (Clarke et al. 2011; Ji et al. 2022). *Alpine-subalpine Closed Heath* is considered **Least Concern,** although we caution that studies of interacting temperature, precipitation and post-fire recovery suggest that flora may respond to fire in non-linear, multidirectional ways that are difficult to predict (Hickman et al. 2024).

Table 17. The number of simulations (out of 100), where at least 1 grid cell of *Alpine-subalpine Closed Heath*, burnt at a frequency that met or exceeded the collapse threshold (assuming 3 fires in 20 years was representative of the collapse threshold being exceeded.), and the corresponding area (in percentage of cells) of *Alpine-subalpine Closed Heath* area (based on the number of grid cells) that this was the case for.

Number of simulations burnt over threshold	**% of Closed Heath area under scenario CSIRO-R1**	**% of Closed Heath area under scenario CSIRO-R3**	**% of Closed Heath area under scenario ECHAM-R1**	**% of Closed Heath area under scenario ECHAM-R2**
0	98.03%	88.51%	94.91%	85.39%
1	0.07%	0.18%	0.10%	0.15%
2	1.74%	10.03%	4.94%	10.46%
3	-	0.07%	-	-
4	0.15%	1.13%	0.04%	3.23%
5	-	-	-	-
6	-	0.07%	-	0.76%

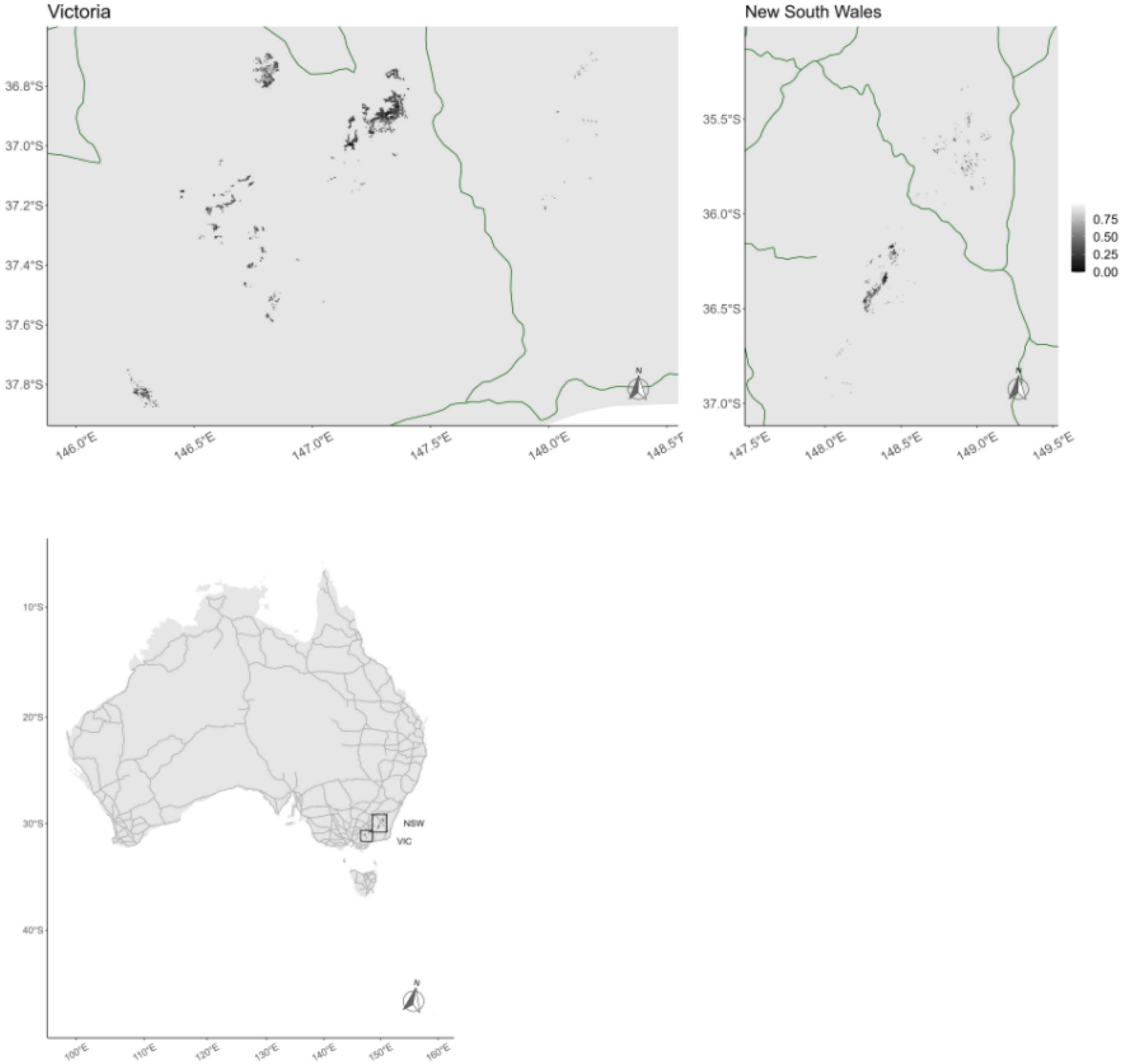


Figure 31. Mean projected future fire frequency (as spatially weighted annual probability of burning) for *Alpine-subalpine Closed Heath* under the ECHAM R-2 scenario. Areas shaded in b Mean is calculated across 100 replicated simulations.

Historical change (C3)

According to fire records, fire frequencies were greatest between ~ 1820 and 1940 with intervals as short as 3.5 – 7.0 years (Zylstra 2006). Following the end of grazing (c. 1960 in NSW), very little of the ecosystem type (< 1 %) was burnt between 1939 -1960. Fires were likely used to clear land in Victorian extents, but this declined during the 1980s (with grazing ceasing in 2003). Since 2000, 87% has burnt at least once (Table 18). However, Australian fire records prior to 1960 are anecdotal in nature; fire extent maps are available for the 1939 Black Saturday fires only. Despite significant declines, flora has exhibited recovery following significant and repeat fire events (e.g., Bogong High Plains, Victoria) (McDougall 2003). From the evidence available, we infer the risk the risk status under sub-criterion C3 is assessed as **Least Concern.**

Table 18. The percentage (%) of the *Alpine-subalpine Closed Heath* area (based on the number of grid cells) that has burnt at different frequencies.

Fire frequency	**Current (2000-2020)**	**Baseline (1939-1960)**
1	64.44%	0.22%
2	20.33%	0.13%
3	2.25%	0
4	0.002%	0

Indicator: Bare ground (%)

Relevance to ecosystem function

Up to 5 % bare ground cover is characteristic of this ecosystem type (Williams et al., 2014). Shrub senescence, fire events, and ungulate trampling and wallowing increase bare ground by removing vegetation and leaf litter (Williams et al. 2014). Following a disturbance, it may take up to a decade for shrub cover and bare ground cover to return to pre-disturbance levels (Williams et al. 2014). Excessive bare ground represents a threat to the *Alpine-subalpine Closed Heath* by increasing the risk of soil erosion and frost heave (Wahren et al. 1994; Williams et al. 2008), depleting the soil seedbank and providing a potential recruitment mechanism for invasive plants.

Data availability and quality

Recent change (C1)

For Victoria, we collated two timeseries of bare ground cover following the 2003 fire in Bogong High Plains from burnt and unburnt sites (Table 19). All sites had experienced historical fires and livestock grazing. While bare ground cover was not explicitly measured, we assumed that the proportion of a quadrat without vegetation was bare ground. Data of bare ground cover between 2003 - 2013 in burnt and unburnt sites were also available for alpine heathland ecosystems in the Bogong High Plains, Victoria (Camac et al. 2015). While not exclusive to *Alpine-subalpine Closed Heath*, these data demonstrate patterns that are indicative of *Alpine-subalpine Closed Heath*.

Data from NSW, the ACT, and the wider Victorian distribution of *Alpine-subalpine Closed Heath* were unavailable at the time of assessment. The disturbance history and species composition of ecosystem distribution within the Bogong High Plains may differ from other ecosystem distribution locations. However, in the absence of other available data, we assumed that the Bogong High Plains ecosystem distribution is broadly representative of changes across the entire ecosystem extent.

Table 19. Estimates of bare ground (%) and shrub cover (%) from field monitoring studies at Bogong High Plains, Victoria. Values are the mean cover (%) ± 95% confidence intervals. These differ in site dimensions and levels of variability. All sites were long-term experimental or monitoring sites. The Bogong High Plains was partially burnt in 2003. Data from Williams & Ashton (1987) pertains to *Alpine-subalpine Closed Heath*. Data from Camac et al. (2013) pertains to alpine heathland, however, trends observed are applicable and inclusive of shrub and bare ground cover changes in *Alpine-subalpine Closed Heath*.

Year	**Bare ground (%)**	**Shrub cover (%)**	**N**	**Study design**	**Citation**
1980	2	NA	5	1.0 m x 1.0 m plots. No evidence of grazing. Sites previously burnt.	(Williams & Ashton 1987)
1983	3	NA	5		
2008	2 ± 0.4 (unburnt 2003)	81.5 ± 3.1	10	50 m transect, 6.0 m^2 at 10 m intervals. Grazing until 2003 in all sites.	(Camac et al. 2013)
2008	16.2 ± 1.6 (burnt 2003)	25.3 ± 2.7	30		

Future change (C2)

Future estimates of bare ground cover were inferred based on published literature of typically cover in the absence and presence of fire and typical recovery rates.

Historical change (C3)

Historical estimates of bare ground cover were inferred based on published literature of typically cover in the absence and presence of fire and typical recovery rates.

Selection of collapse threshold

Bare ground cover is generally low (< 5 %) (Camac et al. 2013; Williams et al. 2014) in undisturbed *Alpine-subalpine Closed Heath.* We therefore assumed that the ecosystem would collapse if bare ground cover was ≥ 10% five years after a disturbance, and shrub cover was < 50 % 10 years after a disturbance and did not show signs of recovery (shrub cover is assessed in Criterion D).

Calculation of severity and extent

Recent change (C1)

We assumed an absence of bare ground represented a relative severity of 0% and presence of bare ground represented a relative severity of 100%. Data suggest that bare ground increases immediately after a fire and is typically low in unburnt sites, even those disturbed by grazing (Camac et al. 2013). These data are likely to represent an upper estimate of bare ground cover across the ecosystem type, as the Victorian range has experienced more fire events than the NSW and ACT distributions. Bare ground cover is unlikely to have increased across the whole extent to meet the threshold for Vulnerable (> 30 % extent), therefore the risk status is **Least Concern** for sub-criterion C1.

Future change (C2)

Future values of bare ground are likely to be low in the absence of disturbances (Armstrong et al. 2013; Wahren et al. 2013). More frequent fires coupled with severe drought are likely to increase bare ground in the short term (< 3 years) (Wahren et al. 2013; Camac et al. 2017), but enhanced shrub growth and shrub thickening under the warming climate will likely counteract this long-term (10 years for shrub coverage to return to pre-fire levels) (Camac et al. 2017). Therefore, risk status is **Least Concern** is under sub-criterion C2.

Historical change (C3)

We would expect the average bare ground cover to be < 5% in the absence of fire and other disturbances (e.g., land use change) (Williams & Ashton 1987; Scherrer & Pickering 2005; Zylstra 2006). Bare ground cover persisting above 3% is rare across the alpine zone (*pers. comm* James Camac, 2021), although has likely increased over this timeframe due to pastoralism (i.e., grazing, fire), warmer, drier conditions (i.e., drought, fire) and an increase in dry lightning (Zylstra 2006; Fraser et al. 2016; Ji et al. 2022). Trampling and grazing by livestock (c. 1840 – 1950 in NSW, ceasing 2003 in Victoria) and feral ungulates has increased bare ground cover at a local scale (Williams et al. 2014). Several fires have increased bare ground cover, followed by decline to pre-fire levels within 10 years (Williams et al. 2014). The ecosystem type is unlikely to meet the threshold for Vulnerable (≥ 50% change), therefore the risk status is **Least Concern** under sub-criterion C3.

Indicator: climate-based indicators

Relevance to ecosystem function

Precipitation as rainfall and snowfall provides an important source of groundwater recharge for *Alpine-subalpine Closed Heath* (McDougall et al. 2015). Snowfall during winter acts as an insulating layer, preventing freezing of the vegetation, soil mineral layer and soil fauna (Green & Osborne 1994). However, climate change is likely to decrease snowfall, induce earlier snowmelt, and increase frost in the absence of a protective snow cover (Sánchez-Bayo & Green 2013). This is most likely to exacerbate soil erosion, soil freezing and frost heave in areas of disturbed *Alpine-subalpine Closed Heath* and increase shrub mortality. Conversely, warmer temperatures coupled with declines in snow coverage appear to promote shrub encroachment into neighbouring ecosystems, seedling germination and survival and thus reduced snowfall is likely to increase the distribution of *Alpine-subalpine Closed Heath* (Venn et al. 2013, 2021; Camac et al. 2021; Sumner & Venn 2022).

Data availability and quality

Snow depth, persistence and frost days provide suitable indicators of snow, snowmelt, and frost frequency respectively. The link between these climatic conditions and ecosystem collapse thresholds is unclear, although ecosystem persistence is not reliant on snowfall (c.f., *Snowpatch Herbfield*) (Williams et al. 2015). Indirectly, declines may occur if soil water is not recharged sufficiently during snowmelt to support persistence of shrubs (i.e., drought) (Sumner & Venn 2022). Rather, changes occurring at the microsite level – reduced snowfall leading to warmer microclimates – are driving ecosystem changes (Green & Pickering 2009a; Lim et al. 2017). Warmer temperatures are driving range expansion and thickening of shrubs and trees throughout the Australina alpine region (Verrall 2023). Local declines due to frost damage before frost hardening may occur (Venn et al. 2013), but whether the frequency and / or severity of future frost events will cause widespread ecosystem declines is not known.

In-situ observational and field experiments over the 30 years (e.g., Wahren et al. (2013) corroborate with long-term, observed changes in shrub cover alongside warmer, drier more disturbance-prone conditions (earliest monitoring c. 1930s, Maisy's plots and aerial photography from the Bogong High Plains, Victoria) (McDougall 2003; Morgan & Green 2013). Observational temperature and preciptiation records are limited to more populated extents of the Australian Alps (e.g., ski resorts such as Falls Creek, Victoria [1990 - present]) (BOM 2025) and current modelled temperature and precipitation datasets (e.g., World Bio Clim) are constrained to 1 km^2 resolution (Fick & Hijmans 2017). Whilst ecosystem declines due to climate change are unlikely, in the absence of comprehensive data to discern quantitativew collapse thresholds in precipitation, temperature and snowfall, the ecosystem is considered **Data Deficient** using these indicators.

Criterion D: Disruption of biotic processes and/or interactions

Summary

Alpine-subalpine Closed Heath is assessed as **Least Concern** under sub-criterion D.

Identification of biotic indicators

We examined the relevance and availability of data for three indicators to assess the risk of collapse from disruption to biotic processes and/or interactions:

- *Shrub cover:* the percentage of overlapping cover of woody shrub vegetation 0.5-2.0 m high.
- *Abundance of exotic herbivores:* the abundance of exotic herbivores in the ecosystem type.
- *Grass and forb cover:* the percentage of overlapping cover of grasses and forbs.

Indicator: Shrub cover

Relevance to ecosystem function

A closed shrub canopy (≥ 70 % cover) is the defining characteristic of *Alpine-subalpine Closed Heath* (Camac et al. 2013). Disturbances such as fire, disease and shrub senescence may lead to short-term declines in shrub cover. However, shrub recovery (via germination or resprouting following fire) occurs within a few weeks of a disturbance, and a dense, closed canopy re-establishing within a decade of the disturbance event.

Data availability and quality

Recent change (D1)

We used data on shrub cover from long-term monitoring sites in the Bogong High Plains, Victoria (Table 20). Williams et al. (2008) dataset is not exclusive to but is indicative of patterns in *Alpine-subalpine Closed Heath.* In addition, we used an aerial photography analysis of *Alpine-subalpine Closed Heath* expansion between 1936 and 1980 (McDougall 2003). Other literature inclusive of shrub coverage from distributions in NSW, ACT and Victoria did not explicitly define the study unit with sufficient detail to align with *Alpine -subalpine Closed Heath* and were omitted.

Future change (D2a)

Quantitative projections of shrub cover change across the entire ecosystem extent were not available. Instead, we used the available evidence from observational and experimental studies of the Australian alpine region, and studies that assessed likely future alpine shrub cover change at a global scale.

Historical change (D3)

We inferred change since 1750 based on studies of shrub cover over the last century (Table 20) and expert knowledge based on personal experience working in the Australian alpine region.

Table 20. Estimates of shrub cover (%) and annual rate of change from field monitoring and aerial photography studies in the Bogong High Plains, Victoria for study sites identified as "closed heathland". Values are the mean % cover (%) ± 95% confidence intervals (CI). N indicates the number of field study sites sampled in the study. All sites were long-term experimental or monitoring sites but differ in dimensions of field site sampled and variability. The Bogong High Plains, Victoria was burnt in 2003 and again in 2006. Data from Camac et al. (2013) aligns with the definition of *Alpine-subalpine Closed Heath.* Data from Williams et al. (2008) pertains to shrub cover in alpine heathland, inclusive of both *Alpine-subalpine Closed Heath and Alpine-subalpine Open Grassy Heathland and Herbfield.*

Year	**Years since fire**	**Shrub cover (%)**	**Change per year (%)**	**CI**	**N**	**Study design**	**Citation**
2006	+3 years	32% (burnt)	+ 10.7%	± 5.7%	3	0.5 ha sample plots. Burnt site sampled in 2008 was burnt in 2003.	(Williams et al. 2008)
2006	-	85% (unburnt)	Stable	± 8.6%	3		
2008[c]	+5 years	25.3% (burnt)	+ 5.0%	± 2.7%	30	50 m long transect, 6.0 m x 6.0 m at 10 m intervals. Burnt or unburnt in 2003 fires. History of grazing until 2003.	(Camac et al. 2013)
2008[c]	-	81.5% (unburnt)	Stable	± 4.4%	10		

Selection of collapse threshold

Shrub cover is high (≥ 70 %) in undisturbed *Alpine-subalpine Closed Heath* (Camac et al. 2013). We therefore defined collapse as when shrub cover is < 50% (Lindenmayer et al. 2014) for 10 years (10 - 15 years) following a disturbance, and there is no sign of recovery. This would represent a transition to *Alpine-subalpine Open Grassy Heathland and Herbfield* as grasses, forbs and herbs increase in cover.

Calculation of severity and extent

Recent change (D1)

Regional spatial analysis suggests that shrubs are encroaching into higher elevation regions, although this upward movement may be offset by tree line advance at lower elevations on the mainland (Verrall 2023). Aerial photography analysis of vegetation change suggests that following the 1939 fire, *Alpine-subalpine Closed Heath* recovered (81% shrub cover) with existing patches expanding from 4.3 % - 8.0 % of area sampled. These increases offset minor transitions of the ecosystem type to *Alpine-subalpine Open Grassy Heathland and Herbfield* (32 % of area) and wetland-type vegetation (2 %, likely recovery following the end of grazing) (McDougall 2003). Evidence from Williams et al. (2008) and Camac et al. (2013) suggests that the ecosystem type is recovering from more recent 2003 and 2006 fires. Assuming linear recovery rates these ecosystems would have reached 50 % shrub cover within 10 years (Table 20). Given the available evidence, we consider that it is unlikely that shrub cover has decreased sufficiently to meet the threshold for Vulnerable. Therefore, the ecosystem type is **Least Concern** under criterion D1.

Future change (D2a)

In the absence of future disturbances, shrub cover is likely to remain high and may even expand into neighbouring *Alpine Open Grassy Heathland and Herbfield* (Armstrong et al. 2013; Wahren et al. 2013; Camac et al. 2017). Results from field-based experimental warming studies suggest that temperature will drive increases in shrub seedling growth rates (Williams et al. 2014; Camac et al. 2017). Most experts consulted agreed that shrub cover is likely to increase by 2050 (Camac et al. 2021). Shrub encroachment into adjacent *Snowpatch Herbfield* has already been observed, as warmer climates reduce snowfall and result in earlier spring snowmelt (Morgan & Walker 2023). Given that shrub cover is likely to increase under a warming climate, the risk status is **Least Concern** under sub-criterion D2a.

Historical change (D3)

Historical information is patchy, although we expect shrub cover to be ≥70% across the ecosystem type in the absence of a disturbance. It was deemed highly unlikely that shrub cover would have declined by enough to meet the threshold for Vulnerable (≥ 50% loss), therefore the ecosystem type is **Least Concern** under criterion D3.

Indicator: Abundance of exotic fauna and native herbivores

Relevance to ecosystem function

The impact of exotic herbivores on alpine ecosystems is widely known (see *Threats*). Invasive herbivores graze and trample native vegetation (Green & Osborne 1994; Nimmo & Miller 2007), exacerbate bare ground (Williams et al. 2014), spread invasive plant species. Exotic ungulates have already driven declines in adjacent ecosystem types such as *Alpine and subalpine bogs and associated fens*, which are also characterised by a shrub flora (Wahren et al. 2001). Herbivores such as deer and rabbits also consume flora, delaying recovery of shrubs from concurrent disturbances such as drought and fire (Leigh et al. 1987). The increased distribution and abundance of deer (already present in adjacent *Alpine and subalpine Open Grassy Heathland and Herbfield*) (Hartley et al. 2022) and pigs (Hone 2002) across *Alpine-subalpine Closed Heath* is likely to exacerbate the abovementioned ecosystem threats and could lead to ecosystem collapse, particularly in the future as warmer temperatures may increase the residence times of invasive fauna in high elevation regions. At present, there is insufficient information on the abundance or distribution of deer and pigs across the alpine region. The abundance of exotic ungulates may be an informative indicator for future assessments, if a minimum density (e.g., individuals per unit area) causing collapse, or other comparable metric can be determined, and if coupled with information on current and predicted distributions to enable quantitative assessment.

Data availability and quality

Evidence of the distribution of exotic ungulates is patchy and constrained to site-specific (largely historical) studies only. In the absence of more comprehensive information, the ecosystem is **Data Deficient** based on this indicator.

Indicator: Native herbivores

Insects such as *Agrotis infusa* (Bogong moth) and *Kosciuscola spp.* (grasshopper) are the dominant alpine herbivores (Balmer 2025, *pers. comm.*; Green 2010; Green & Osborne 2012). *Kosciusko spp.* rely on the presence of an insulating snow layer that accumulates on the top of the closed shrub canopy (subnivean space) to insulate them from the cold climatic conditions. Reductions in snow are likely throughout the Australian alpine region (Hennessy et al. 2008; Sánchez-Bayo & Green 2013) and large populations of these semelparous insects may be wiped out due to exposure to freezing conditions. Populations of migratory *Agrotis infusa* may also decline if disturbances (e.g., drought, high temperatures) disrupt maturation and emergence in non-alpine regions where these insects develop to maturity (Lownds et al. 2023). Phenological mismatches may also occur as the migratory and flowering cues become mismatched, leading to both a loss of food source for fauna and decline in reproductive capacity of shrub biota. Subsequent declines in predatory fauna may also occur (Green & Sanecki 2006). Ecosystem processes associated with both herbivory and pollination will be lost and thus functional collapse may occur.

Data availability and quality

Evidence of pollinator mismatches been documented in literature (e.g., Green & Sanecki 2006), as have population declines associated with exposure to high temperatures at non-alpine aestivation sites (Lownds et al. 2023). However, there is insufficient evidence available to determine quantitative relationships between insect populations, pollination, herbivory and ecosystem collapse. In the absence of more comprehensive information, the ecosystem is **Data Deficient** using this indicator.

Indicator: Grass, forb and tree cover

Relevance to ecosystem function

The loss of the dominant shrub canopy (< 50 % shrub cover 10 years after disturbance) would indicate ecosystem collapse. If grasses and forbs become dominant, this would represents a transition to *Alpine-subalpine Open Grassy Heathland and Herbfield*. If *Eucalyptus spp.* establish (e.g., > 4.0 m tall) this would represent at transition to *Alpine and subalpine Woodlands and Forest* with a closed heathland understorey. The ecosystem may also collapse into theoretical novel state if exotic species become dominant but this pathway is considered unlikely.

Data availability and quality

Evidence from Williams et al. (2008) suggests that the post-fire recovery rates of grasses and forbs are faster than the recovery rates of shrubs (Walsh & Mcdougall 2004; Camac et al. 2013; McDougall et al. 2015; Verrall 2018). Increasing tree cover is most likely to signal ecosystem declines but data available cannot be downscaled to the ecosystem type. At present, these ecosystem collapse pathways are already represented by the indicator of shrub cover although non-shrubby vegetation cover may be an informative indicator for future assessments. Thus, the ecosystem type is considered **Data Deficient** under these indicators.

Criterion E: Quantitative analysis of probability of collapse

Summary

No stochastic models of *Alpine-subalpine Closed Heath* are available and there is presently insufficient data to reliably inform simulations. Therefore, the risk status is **Data Deficient** under criterion E.

Photo: Susanna Venn.

Coniferous Heath

Authors

David Keith, Jamie Kirkpatrick, Jessica Rowland, Ella Plumanns Pouton

Reviewers

Jennie Whinam, Emily Nicholson

Biome

T6 Polar-alpine

Functional group

T6.4 Temperate alpine grasslands and shrublands

IUCN status

Vulnerable (Vulnerable-Endangered)

Podocarpus shrubland at the Cobberas, Victoria. These populations were partly burnt in 2003. Photo: Arn Tolsma.

Assessment Summary

Coniferous Heath is a low, usually dense shrubland dominated by a suite of palaeoendemic coniferous shrubs and sparse emergent coniferous trees. It is unique to the Tasmanian central plateau and southern mountains and in highly restricted rocky sites in the Snowy mountains and Victorian alps. This ecosystem type is typically characterised by a single species of conifer and is habitat for the mainland alpine endemic mountain pygmy possum (*Burramys parvus*). *Coniferous Heath* is threatened by climate change and an associated increase in occurrence and extent of fires, as the dominant plants have no regenerative traits that enable their populations to re-establish when standing plants are killed by fire.

The ecosystem type was assessed as **Vulnerable (Vulnerable-Endangered)** throughout its range due to inferred historic fire-related declines (sub-criterion A3), and a restricted distribution (small number of threat-defined locations) and fire-related threats that may cause the ecosystem type to collapse or become Critically Endangered within a very short period (sub-criterion B3) (Table 21). However, the disjunct occurrence in both Tasmanian and mainland distributions are each likely to meet sub-criterion B1 for **Endangered** status due to their very restricted Extent of Occurrence and plausible threats likely to cause continuing declines (sub-criterion B1).

Table 21. Summary of the Red List of Ecosystems assessment of the Australian *Coniferous Heath.* Category ranges in parentheses reflect uncertainty in assessment under the corresponding criteria.

Criteria	A	B	C	D	E	Overall
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	LC	LC	DD	DD	**VU (VU-EN)**
Sub-criterion 2 *A, C, D: 50-year period including present & future* *B: AOO*	DD	LC	NT	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	VU (VU-EN)	VU	DD	DD		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria

Ecosystem Description

Ecosystem Classification

In the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022a), this ecosystem type belongs to Ecosystem Functional Group *T6.4 Temperate alpine grasslands and shrublands* within the *Polar/alpine (cryogenic) biome*.

Coniferous Heath includes Podocarp Shrubland on the mainland and Coniferous Heath in Tasmania (Kirkpatrick 1983, 1997; Costin et al. 2000; McDougall & Walsh 2007; Venn et al. 2017). In Tasmania, the ecosystem type is mapped as vegetation community HCH (Alpine coniferous heathland) and is distinct from RPW (*Athrotaxis cupressoides* open woodland), which attains woodland structure (> 5% tree cover), and sometimes with other conifers in the understorey (Harris & Kitchener 2005). Risks to *Athrotaxis cupressoides* open woodland are therefore assessed separately as a different ecosystem type. On the mainland, the *Coniferous Heath* ecosystem type includes communities 53 and 54 (boulder heathlands) described by McDougall & Walsh (2007). *Coniferous Heath* is mapped as Ecological Vegetation Class 156 (Alpine Coniferous Shrubland) in Victoria (Mackey et al. 2015). *Coniferous Heath* has not been explicitly mapped in NSW, but is closely associated with block streams. Mapping of block streams as *Burramys parvus* habitat (Broome et al. 2013) therefore serves as surrogate spatial data representing the distribution of *Coniferous Heath* in NSW.

Although not currently listed under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC), these vegetation types are within National Estate Heritage areas and World Heritage areas and therefore considered as 'Matters of National Environmental Significance' under the EPBC Act.

Distinction from similar ecosystem types

Coniferous Heath is similar in structure *to Alpine-subalpine Closed Heath* (found on the mainland only) and *Tasmanian Alpine Heath*. *Coniferous Heath* is dominated by long-lived palaeoendemic coniferous shrubs that have no regenerative or reproductive traits that enable their persistence through fires. In contrast, *Alpine-subalpine Closed Heath* is defined by a dense canopy of scleromorphic shrubs and *Tasmanian Alpine Heath* is dominated by a diverse assemblage of largely Tasmanian endemic scleromorphic shrubs, equipped with either regenerative organs or persistent soil seedbanks, as well as tolerance of exposure to strong winter winds.

Distribution

Coniferous Heath is largely confined to the alpine zone of Tasmania. However, one expression of the ecosystem type that is dominated by *Podocarpus lawrencei* (Podocarp shrublands) extends to mainland Australia, where it is restricted to block streams and includes some unique biotic elements (e.g., Burramys parvus). The contemporary distribution of *t*he ecosystem type is located between 145.79° to 148.90° longitude and -35.50° and -43.35° latitude.

At local scales, *Coniferous Heath* is usually restricted to small patches. In Tasmania, these patches are scattered on the unburnt western parts of the Central Plateau (Cradle Mountain, Lake St Clair), the West Coast Ranges, the Tyndall Ranges, the Southern Ranges, Mt Field and southwest mountains (Kirkpatrick 1997). The mainland expression of the ecosystem type is scattered on the Bogong High Plains, Mt Hotham, Mt Howitt, Mt Buffalo, Cobberas and Cross Cut Saw in Victoria (McDougall & Walsh 2007); and between Mt Koscuiszko and Mt Jagungal in NSW (McDougall &

Walsh 2007). The total mapped area of *Coniferous Heath* covers approximately 53.72 km^2 (Figure 32).

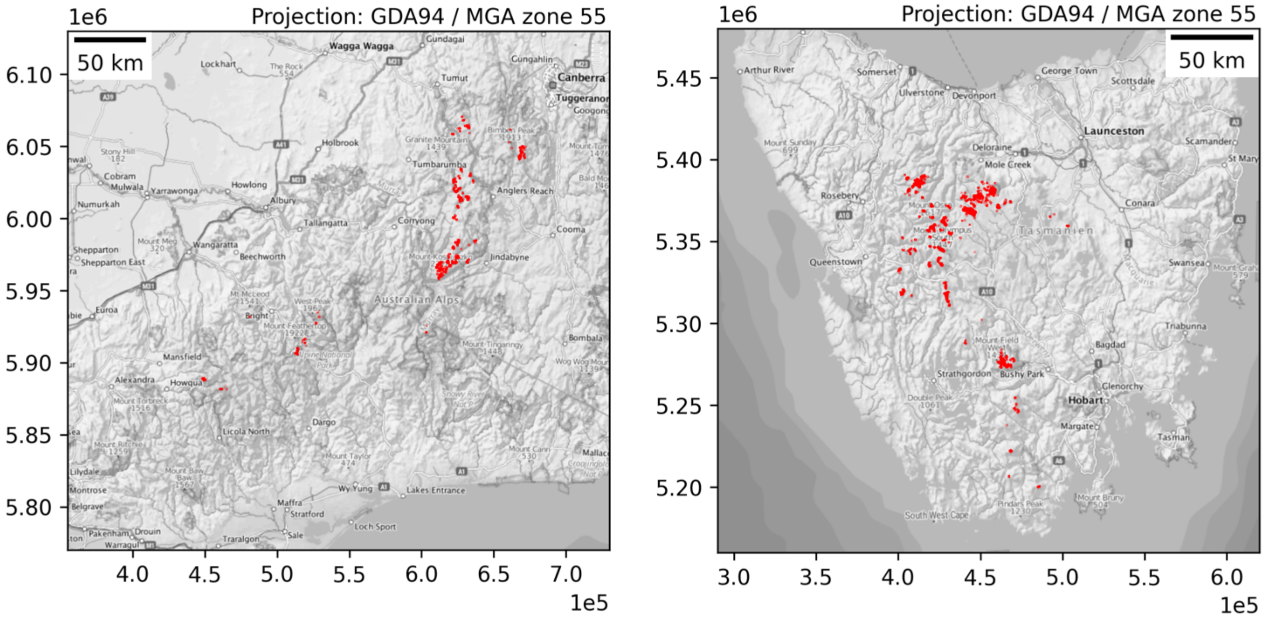


Figure 32. Distribution of *Coniferous Heath* (red) across the Australian mainland (left) and Tasmania (right).

Abiotic environment

Coniferous Heath is largely restricted to climatically wet alpine regions, mainly in Tasmania where mean annual precipitation exceeds 2000 mm per annum. Small patches occur in protected sites where rainfall may be as low as 1200 mm per annum (e.g., Mt Wellington). Snow cover is not reliable in winter, exposing the dominant shrubs to wind abrasion and low temperatures. Winters are typically cold (mean temperature of the coldest quarter is -2.0°C) and summers cool (mean temperature of warmest month < 10°C), with a low annual temperature range. They occupy a wide range of substrates that produce acidic–neutral soils with varied nutrient status. They are closely associated with fire refuges because of either continually high climatic moisture, topographic shelter (e.g., southern mountain slopes), rocky microhabitats (e.g., block streams, screes), water bodies (on islands or isthmus), snow patches or some combination of those features (Kirkpatrick 1997; Venn et al. 2017). On the mainland, they are almost entirely restricted to block streams.

Characteristic native biota

Coniferous Heath is dominated by a suite of palaeoendemic coniferous shrubs – primarily *Microcachrys tetragona, Pherosphaera hookeriana*, *Diselma archeri* and *Podocarpus lawrencei* (Figure 33) (Kirkpatrick 1997). Other conifers may occur as dwarf forms or as scattered emergent trees, particularly *Athrotaxis cupressoides*, but also *A. selaginoides* and *Lagarostrobos franklinii*. Different combinations of these plant species may co-occur and dominate, depending on local environmental conditions. *Diselma archeri, Microcachrys tetragona* and the more restricted *Pherosphaera hookeriana* occur in locally wet sites, such as those associated with peat ponds or snow patches. They may also extend to more freely draining sites. Dwarf *Arthrotaxis cupressoides* dominates frost-prone flat terrain, but this species also occurs as emergent trees in rocky sites. *Podocarpus lawrencei* dominates rocky screes and block streams and is the only alpine conifer species that extends to mainland Australia. Dense nanophyllous foliage, well-protected buds and slow growth rates potentially favour persistence of these species in exposed cold winter and spring-thaw conditions, likely conferring a fitness advantage over sclerophyllous angiosperms that dominate other heathland ecosystems.

Coniferous Heath shares several plant and animal species with adjacent *Alpine-subalpine Closed Health, Tasmanian Alpine Heathland, Cushion Moorland* or *Snowpatch Herbfields*, including *Abrotanella forsteroides*, *Richea sprengelioides*, *Trochocapra cunninghamii*, *Orites revoluta*, *Ozothamnus rodwayi* in Tasmania, and *Epacris paludosa* and *Poa gunnii* both in Tasmania and on the mainland (Harris & Kitchener 2005; McDougall & Walsh 2007).

Characteristic vertebrates in this ecosystem type include small mammals such as *Antechinus swainsonii, Pseudomys fuscus* and *P. higginsi*, and endemic skinks, including several *Niveoscincus* species in Tasmania, and the Endangered Guthega skink (*Liopholis guthega)* on the mainland. Currawongs (*Strepera* spp.), wombats (*Vombatus ursinus*) and wallabies (*Macropus rufogriseus, Wallabia bicolor*) disperse from adjacent lowland populations. *Coniferous Heath* on the mainland is primary habitat for *Burramys parvus* (mountain pygmy possum) (McDougall & Walsh 2007). The possum's shelter in block streams and feed on fruits of *Podocarpus. Agrotis infusa* (the Bogong moth) also shelters amongst the boulders. Important invertebrate groups include dipterans, lepidopterans and burrowing crustacea. The invertebrate fauna is not species-rich but includes several local endemics.

Figure 33. Coniferous Heath on Mt Field, Tasmania. Photo: Jamie Kirkpatrick.

Key components, processes, and interactions

Fire refugia

The standing plants of palaeoendemic conifers lack fire-resistant organs and have no seed banks to support recruitment after fire. They have slow growth rates, irregular cone production, short propagule dispersal distances and low rates of recruitment (e.g., Gibson et al. 1995). Fires that burn all individuals and thus eliminate populations from a site. Re-establishment generally depends on recolonisation from unburnt sources, a process that may take centuries or millennia due to relatively localised dispersal mechanisms (Kirkpatrick et al. 2010; Fletcher et al. 2014). The only alpine conifer on the mainland, *Podocarpus lawrencei*, occasionally shows weak resprouting responses after low severity scorching, but only some resprouting individuals survive the post-fire years (Walsh & Mcdougall 2004). Post-fire seedling recruitment appears rare but has been observed at one site (McDougall & Walsh 2007).

Consequently, this ecosystem type is restricted to fire refugia (Figure 34). These are sites where fire spread may be inhibited by any of several mechanisms including: moist microclimates created by topographic shelter (e.g., southern aspects); fuel discontinuities created by water bodies (islands, isthmus); snow patches; rocky areas (block streams, screes); or sites that receive very high precipitation and have low evapotranspiration rates (as in the western and southern Tasmanian mountains). In contrast, several other woody plant species that may co-occur with the conifers have persistent soil seedbanks (e.g., *Ericaceae*), fire resistant subterranean recovery organs (e.g., *Orites revoluta*), or effective long-distance dispersal mechanisms enabling post-fire recolonization (e.g., *Asteraceae*). Recurrent fires potentially promote these shrub species over the conifers.

Dispersal

Small mammals, skinks and currawongs play important roles in dispersal of fleshy-fruited plants (Figure 34), including some of the conifers (*Podocarpus*, *Microcachrys*). *Podocarpus* fruits also make up a key dietary component for *Burramys parvus* (mountain pygmy possum) (McDougall & Walsh 2007). Macropod browsers disperse from lowland populations and may limit recruitment in of some woody species, especially *Athrotaxis cupressoides* (Cullen & Kirkpatrick 1988). A range of dipterans function as pollinators and detritivores, while the conifers and other characteristic plant species are specialist larval hosts for some invertebrates (e.g., moths).

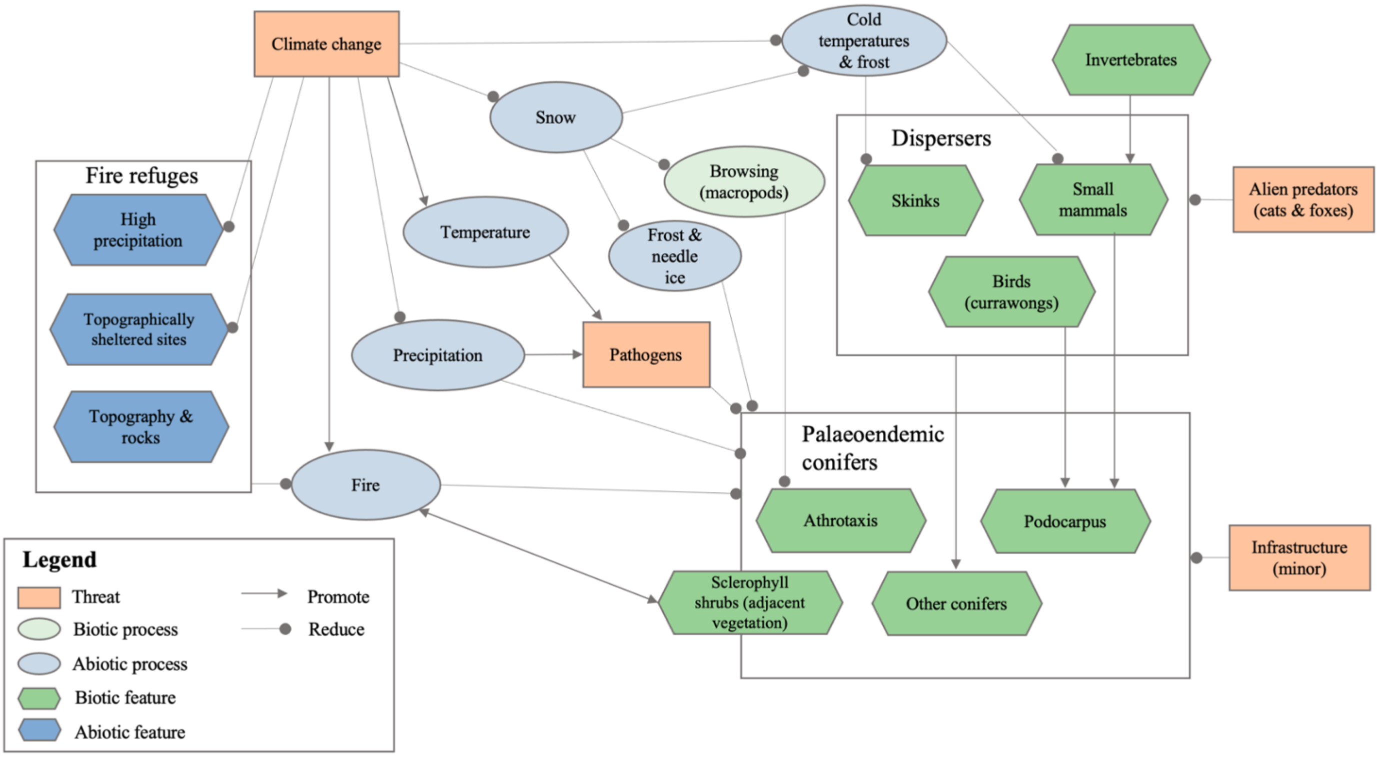


Figure 34. Conceptual diagram illustrating relationships between, and threats to, biotic and abiotic components and processes in *Coniferous Heath*.

Major threats

Changing fire regimes

The interaction of anthropogenic climate change with fire regimes is the primary threat to *Coniferous Heath* (Figure 35). Projections for increased frequency of extreme fire weather, increased severity and duration of droughts (Zylstra 2018; Dowdy et al. 2019), and increased ignitions from lightning (Styger et al. 2018) suggest an increasing risk that fires will transform *Coniferous Heath* to alpine sclerophyll heathlands (*Alpine-subalpine Closed Heath* and *Tasmanian Alpine Heath*) or sedgelands (*Tasmanian Sedgelands*) by eliminating fire-sensitive palaeoendemic conifers. These shrubby conifers in *Coniferous Heath* lack arborescent growth forms and thick bark that may enable some individuals of *Athrotaxis* species to escape mortality where it occurs in burnt stands of *Subalpine Palaeoendemic Forest and Woodland* (Kirkpatrick & Bridle 2013).

Figure 35. Rocky dolerite site with a former patch of *Coniferous Heath* dominated by Podocarpus lawrencei, which was killed by a fire in January 2016. In a sample of 20 plants, none survived the fire, despite rocky microhabitats. Near Lake Mackenzie, Tasmania. Photo: David Keith (November 2019).

Climate change

Climate change may result in reduced snow cover (Sánchez-Bayo & Green 2013), increasing the exposure of seedlings to needle ice and frost and thereby reducing growth, reproduction and survival, and exposing small mammals and reptiles to thermoregulatory stress (OEH 2011). Climatic warming may also enable macropod browsers to inhabit high-elevation *Coniferous Heath* more frequently, increasing browsing pressure on palatable plants, such as *Athrotaxis cupressoides*. Climatic warming may alter reproductive phenologies (Visser & Both 2005; Jarrad et al. 2008; Hoffmann et al. 2010). If this occurs asynchronously within pollination networks, it could reduce reproductive rates of plants and survival rates of their insect pollinators.

Habitat loss and fragmentation

Development of infrastructure related to tourism (ski runs, accommodation and service facilitates) and energy (hydro-power pump stations, pipelines, energy transmission lines, etc.) may result in loss of some patches of *Coniferous Heath*.

Invasive species

Changes in populations of prey (e.g., bogong moths) and feral predators (e.g., cats, foxes) are likely to cause continuing declines in small mammals that inhabit *Coniferous Heath*, most notably the endemic mountain pygmy possum (Hughes 2011; OEH 2011).

IUCN Stresses Classification

1.2 – Ecosystem degradation
1.3 – Indirect ecosystem effects

IUCN Threats Classification

1.3 – Tourism and recreation areas
7.2 – Dams & Water Management/Use
7.1.1 – Increase in fire frequency / intensity
8.1 – Invasive non-native species
11.1 – Habitat shifting and alteration
11.2 – Droughts
11.3 – Temperature extremes

Ecosystem Collapse

Coniferous Heath collapses when the characteristic conifer species lose dominance. This may occur if the ecosystem type is burnt, as a single fire can cause total mortality (Kirkpatrick et al., 2010). The ecosystem type may transition into *Tasmanian Alpine Heath* or *Alpine-subalpine Closed Heath* if sclerophyllous shrubs become dominant, or *Tasmanian Sedgelands* if hard-leaved monocotyledonous plants become dominant.

Coniferous Heath collapses when any of the following occur:

1. *Area*: The mapped distribution declines to zero (100% loss) (Criteria A and B), or;
2. *Fire occurrence*: All patches have been burnt within 200 years, a period expected to be much shorter than that required for post-fire recovery (Criterion C), and/or;
3. *Conifer vegetation cover*: Density of characteristic conifer species is < 10 individuals per 0.1 ha or canopy cover is < 5 % throughout the distribution (Criterion D).

Risk Assessment

Criterion A: Reduction in distribution

Summary

The status of *Coniferous Heath* under criterion A is **Vulnerable (Vulnerable – Endangered)** based on historical declines in the distribution caused by fires (sub-criterion A3). Assessments of sub-criteria A1 and A2 produced outcomes of **Least Concern** and **Data Deficient**, respectively.

Methods

Time-series maps of *Coniferous Heath* are unavailable. However, losses in area can be estimated based on historical records of area burnt and predicted future change in fire and lightning occurrence.

Assessment Outcome

Recent decline (A1)

Decline in the distribution of *Coniferous Heath* over the past 50 years is likely to involve relatively small areas overall. Only a few fires have burnt alpine Tasmania since 1970 (Marsden-Smedley 1998), and these affected relatively small areas of *Coniferous Heath*. Approximately 4.99 km^2 of *Coniferous Heath* has been burnt on the mainland since 1970 (see criterion C), which accounts for a large portion of the mainland extent (86.44%). In combination, these losses are likely to account for substantially less than 30% of the distribution of Coniferous Heath over the past 50 years. The status of the ecosystem type under sub-criterion A1 is therefore **Least Concern**. If the assessments were conducted at the state level, the risk status for the mainland extent would be Critically Endangered.

Future change (A2)

Future declines in the distribution of *Coniferous Heath* depend on fire occurrence and severity. This depends on the probability of ignitions from dry lightning strikes and human sources, the probability of extended dry spells that reduce fuel moisture content prior to ignition, and the probability of severe fire weather events (high wind speeds, high temperatures and low humidity). A full statistical model of these factors has not yet been developed, but Styger et al. (2018) found that the mean area burnt by lightning-ignited fires in Tasmania increased during 2000-2016, after a period from 1980-2000 when very small areas were burnt by lightning-ignited fires. They were unable to attribute the cause of this change due to insufficient data, but likely explanations included an increase in lightning frequency or an increase in fire spread from lightning strikes. The 2019-2020 fire season suggests continuation of this trend, as fires burnt mainland alpine ecosystems, after an extended period of low fuel moisture coincident with montane lightning strikes. Climate models project slight increases in dry lightning and stronger tendencies towards drier summers (Love et al. 2016; Styger et al. 2018). However, there are insufficient data to project the rate of fire-related contraction of *Coniferous Heath*. Its status under sub-criterion A2 is therefore **Data Deficient**.

Historical change (A3)

Present day mapping products suggest the ecosystem type is 53.72km^2, 47.38km^2 of which is in Tasmania. The post-1750 industrial era reached Tasmania soon after 1803 when European people invaded and settled on the island. The European history of the mainland alpine region began slightly later, after c. 1820. Historical declines in distribution and function are therefore estimated since that time.

The historical distribution of *Coniferous Heath* in the early 19th century is poorly known, but very likely included areas in Tasmania that were transformed into sclerophyll heathlands (*Tasmanian Alpine Heath* or *Alpine-subalpine Closed Heath*) and *Tasmanian Alpine Sedgeland* when they were burnt in major fires (Kirkpatrick & Bridle 2013). These areas are now mapped as *Tasmanian Alpine Heath* (map units HHE and HHW in TASVEG 4.0) and *Tasmanian Alpine Sedgeland* (map units HSE, HSW in TASVEG 4.0). The major fires that caused collapse of these areas of *Coniferous Heath* occurred primarily in the 1890s, 1930s and, to a lesser degree, the 1950s (Marsden-Smedley 1998). Similar transformations may have occurred in mainland *Coniferous Heath* of NSW and Victoria, where major fires occurred in the summers of 1938-39, 2002-03, 2006-07 and 2019-20 (Walsh & Mcdougall 2004; Keith et al. 2022b).

Based on the most recent mapping (TASVEG 4.0) (DPIPWE 2020), the area of alpine sclerophyll heaths (map units HHE and HHW) (*Tasmanian Alpine Heath* or *Alpine-subalpine Closed Heath*) is 869.13 km^2 and the area of *Tasmanian Alpine Sedgeland* is 396.27 km^2. A conservative assumption that 5-10% of present-day *Tasmanian Alpine Heath* and *Tasmanian Alpine Sedgeland* had been *Coniferous Heath* transformed by fire since European settlement produces an estimated historical decline as low as 59% (100 × (1 - 439 / (439 + 0.5 × (869.13 + 396.37)))) or as high as 74% (100 × (1 - 439 / (439 + 0.5 × (869.13 + 396.37)))). The proportional declines could be similar on the mainland, but even if the ecosystem underwent no decline in distribution on the mainland, the plausible bounds of decline are 59-74% because the estimated area of *Coniferous Heath* on the mainland covers 89% of the distribution. The plausible estimates of decline in distribution of *Coniferous Heath since 1750* span a range of 59-74% (midpoint 66.5%). The status of the ecosystem type under sub-criterion A3 is therefore **Vulnerable (Vulnerable – Endangered).**

Criterion B: Restricted distribution

Summary

The status of *Coniferous Heath* under criterion B is **Vulnerable**, due to sub-criterion B3. The ecosystem type is Least Concern under sub-criterion B1 and B2.

Methods

Under criterion B, the extent of occurrence (EOO; sub-criterion B1) and area of occupancy (AOO; sub-criterion B2) *of Coniferous Heath* were determined using a combination of existing map products from the Australian Capital Territory (ACT Government 2018), New South Wales (Broome et al. 2013), Victoria (Heinze & Harvey 2006; Harvey 2007; DELWP 2016, 2021), and Tasmania (Kitchener & Harris 2013; DPIPWE 2020). A complete description of the methods used to create the current distribution map for this ecosystem type is provided in the main methods.

The number of threat-defined locations was based on the potential extent of fires as this is the most important stochastic threat to Co*niferous Heath* (sub-criterion B3).

Assessment Outcome

Sub-criterion B1

A minimum convex polygon enclosing all mapped occurrences of *Coniferous Heath* in Tasmania and on mainland Australia has an area of 116,600.76 km^2, substantially larger than the threshold Extent of Occurrence values for the Vulnerable category. The status of the ecosystem type across both regions under sub-criterion B1 is **Least Concern**.

Sub-criterion B1

The combined distribution of *Coniferous Heath* in Tasmania and on mainland Australia intersects 79 10×10 km grid cells (Figure 36), which exceeds the Area of Occupancy threshold value for the Vulnerable category. The status of the ecosystem type across both regions under sub-criterion B2 is therefore **Least Concern**. If the Tasmanian and mainland subtypes are assessed separately, they occupy 49 and 30 10×10 km grid cells, respectively, both within the threshold for Vulnerable under sub-criterion B2.

Sub-criterion B3

Based on fire as the most serious plausible threat to *Coniferous Heath*, the ecosystem type is estimated to occupy 2-10 threat-defined locations. A fire in 1898 burnt across the entire distribution of *Coniferous Heath* in Tasmania but left some patches unburnt. Similarly, a fire in 1939 burnt across the entire distribution of *Coniferous Heath* in the mainland Australian alps, but also left some patches unburnt (Fire Information Systems Group 2008). These precedents suggest that the entire distribution could be affected by a small number of fire events. Fires could therefore cause the ecosystem type to collapse or become Critically Endangered within a very short period (c. 20 years). It therefore meets the requirements for **Vulnerable** status under sub-criterion B3.

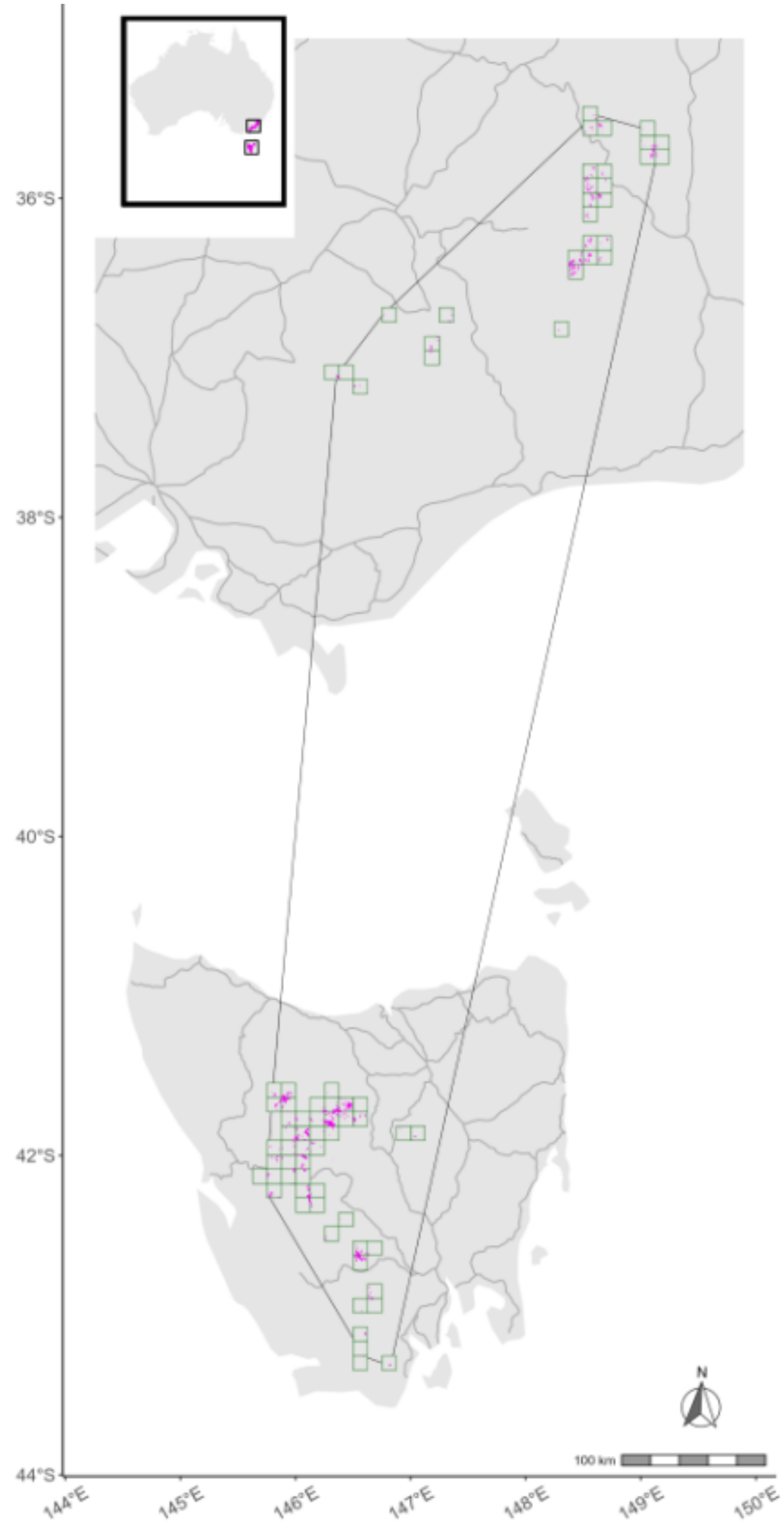


Figure 36. Map of Coniferous Heath (magenta polygons) across Australia, showing Extent of Occurrence (black polygon) and Area of Occupancy where the 1% rule was not applied (green squares).

Criterion C: Environmental degradation

Summary

This ecosystem type is classified as **Least Concern** under sub-criterion C1, **Near Threatened** under sub-criterion C2, and **Data Deficient** under sub-criterion C3 using an indicator of fire occurrence.

Identification of abiotic indicators

We examined the relevance and data availability/quality for one potential indicator to assess the risk of collapse from environmental degradation:

- *Fire occurrence*: a direct measure of fire incidence in *Coniferous Heath*.

Indicator: Fire occurrence

Relevance to ecosystem function

Fire has been a historically rare event in *Coniferous Heath* as it is found on long-unburnt slopes (Venn et al. 2017). However, climate change is altering fire regimes in alpine and subalpine ecosystems across Australia (Zylstra 2018); the frequency of fires is projected to rise due to increases in the frequency of extreme fire weather, the severity and duration of droughts (Zylstra 2018; Dowdy et al. 2019), and the frequency of ignitions from lightning (Styger et al. 2018). The *Coniferous Heath* vegetation comprises non-arborescent growth forms, and its component species lack thick bark and other organs that might protect them from fire damage and mortality (Kirkpatrick & Bridle, 2013). The coniferous species also lack seedbanks, and post-fire recruitment is rare (McDougall & Walsh, 2007). One alpine conifer species on mainland Australia, *Podocarpus lawrencei,* has shown minor resprouting after a low-severity burn, but few of these individuals survived in the years after the fire (Walsh & Mcdougall 2004). Elimination of the conifers by fire is likely to initiate a transition to other treeless ecosystem types. Therefore, *Coniferous Heath* is fire-sensitive and a single fire event can cause the ecosystem type to transform into alpine sclerophyll heaths (on the mainland or Tasmania; *Tasmanian Alpine Heath* or *Alpine-subalpine Closed Heath*) or *Tasmanian Alpine Sedgeland* (in Tasmania).

Data availability and quality

Recent change (C1) and Historical change (C3)

Historical records of bushfires and planned burns are available as shapefiles, detailed in the main methods.

Future (C2a)

We obtained future fire simulations for mainland Australian alpine and subalpine ecosystems for 2060 to 2079 from the landscape fire modelling framework 'Fire Regime and Operations Tool' (FROST), detailed in the main methods.

Selection of collapse threshold

A single fire can eliminate *Coniferous Heath* for centuries or millennia, as growth rates are slow and dispersal distances are short (Kirkpatrick et al., 2010). An optimistic assumption would be that burnt *Coniferous Heath* could recover within 200 years. Therefore, as a very conservative estimate, we assumed the ecosystem type would collapse when all patches have been burnt within a 200-year period (or since the industrial era).

Calculation of severity and extent

Recent change (C1)

Fires have burnt *Coniferous Heath* in 23 fire seasons between 1970 and 2020, covering a total of 4.99 km^2 (9.29 % of the distribution), including 4.89 km^2 on the mainland, and 0.1 km^2 in Tasmania. The relative severity of ecosystem area burnt is therefore estimated to be (4.99/53.14) × 100 = 9.39%. The ecosystem type's risk status is therefore **Least Concern** under sub-criterion C1.

Future change (C2)

Under all four climate scenarios, there were low predicted fire frequency, however the mean predicted fire frequency reached the collapse threshold (1 fire; ECHAM_R2) for 17% of the mainland extent. Overall, the probability of any part of the ecosystem type (i.e., at least one grid cell) burning at a frequency that met or exceeded the collapse threshold ranged from 18% (CSIRO-R1) to 34% (ECHAM R1, ECHAM R2) (Table 22). The extent of the ecosystem type that exceeded the collapse threshold (i.e., relative severity = 100%) in one or more simulations ranged from 48.22% (CSIRO-R1) to 62.2% (ECHAM-R2). Only very small parts of the mainland extent (0.12%) displayed a 34% probability of collapse. However, under ECHAM R2, most of the mainland extent (62.2%) displayed at least some probability of collapse, but this is predominantly a probability between 1 and 14%. Given this, and the likelihood that this analysis is also applicable to the Tasmanian extent of the ecosystem type, relative severity is 18-34% over 48.22-62.2% of the distribution. The risk status is **Least Concern** under sub-criterion C2.

Historical change (C3)

Records estimate that 5.18 km^2 (9.74%) of the ecosystem type has been burnt between 1939 and 2020, mostly in 2002-2003 on mainland Australia (Walsh & Mcdougall 2004) and in the 1890s, 1930s and, to a lesser degree, the 1950s in Tasmania (Marsden-Smedley 1998). Losses of the ecosystem type from fires since 1820 are likely to account for substantially less than 50% of the ecosystem type's distribution. However, fire records prior to 1960 are unreliable. Therefore, the risk status under sub-criterion C3 is assessed as **Data Deficient.**

Table 22. The percentage (%) of the ecosystem type area (based on the number of grid cells) and the number of simulations (out of 100) that burnt at a frequency that met or exceeded the collapse threshold for each climate scenario.

Number of simulations	**CSIRO-R1**	**CSIRO-R3**	**ECHAM-R1**	**ECHAM-R2**
0	61.78	45.72%	44.74	37.80
1	0.73	1.03	0.85	1.10
2	20.21	14.00	17.10	14.49
3	0.43	1.03	0.06	0.18
4	9.31	11.87	11.56	11.69
5	0.06	0.49	0.24	0.18
6	3.77	7.85	7.85	10.83
7	0.06	0.37	0.18	0.24
8	2.13	5.78	4.32	8.58
9	0.06	0.12	0.06	0.18
10	0.85	4.93	2.86	4.14
11		0.06	0.06	1.40
12	0.30	3.23	2.31	1.40
13		0.06	0.06	
14		1.58	1.70	2.37
15		0.18		0.06
16	0.24	1.52	1.34	1.77
17		0.18	0.06	
18	0.06	0.30	1.34	1.70
19				
20		0.24	0.85	1.10
21		0.06		
22		0.24	1.10	0.97
23		0.06		
24		0.06	0.49	0.49
25		0.06		
26		0.06	0.49	0.30
27				
28			0.12	0.12
29				
30			0.06	0.12
31				
32			0.06	0.12
33				
34			0.12	0.12

Criterion D: Disruption of biotic processes and/or interactions

Summary

The dominant palaeoendemic shrubs of *Coniferous Heath* may be susceptible to mortality and declines in density in response to heatwaves and droughts. Reductions in snow cover can reduce growth and survival of seedlings due to exposure to needle ice and frost, whereas warming may enhance browsing by macropods and alter reproductive phonologies and ultimately reduce reproductive rates and result in declines in vegetation cover. However, insufficient data are available to assess this process, and hence the status of the ecosystem type is **Data Deficient** under criterion D.

Identification of biotic indicators

We examined the relevance and data availability/quality for one indicator to assess the risk of collapse from disruption of biotic processes and/or interactions:

- *Conifer vegetation cover*: a direct measure of density of the characteristic conifer species.

Indicator: Conifer vegetation cover

Relevance to ecosystem function

Coniferous Heath is characterised by palaeoendemic conifer shrubs, primarily *Microcachrys tetragona*, *Pherosphaera hookeriana*, *Diselma archeri* and *Podocarpus lawrencei* (Kirkpatrick 1997). The loss of the coniferous shrubs would lead to the loss of the ecosystem type.

Selection of collapse threshold

Conifer vegetation cover in *Coniferous Heath* can vary from 40-60% (Kirkpatrick et al. 2010). Therefore, we consider the ecosystem type to collapse when the density of the characteristic conifer species to be < 10 individuals per 0.1 ha, or when the canopy cover is < 5% throughout the distribution.

Data availability and quality

Insufficient data are available to assess indicator.

Criterion E: Quantitative analysis of probability of collapse

Summary

Insufficient data are available to estimate the probability that *Coniferous Heath* will collapse within the next 50 to 100 years. Hence the ecosystem type is **Data Deficient** under criterion E.

Fjaeldmark/feldmark

Authors

Jessica A. Rowland, Brodie Verrall, Emma Sumner, Susanna Venn, Jamie Kirkpatrick

Reviewers

Jennie Whinam, Clare Vernon, Emma Sumner

Contributors

Matt White, Michael Traurig, Catherine Pickering, Ken Green, Casey Gibson

Biome

T6 Polar-alpine

Functional group

T6.4 Temperate alpine grasslands and shrublands

IUCN status

Vulnerable

Assessment Summary

Australian *Fjaeldmark* is restricted to high elevation, exposed ridgelines and summits of New South Wales and Tasmania, where the vegetation is exposed to periglacial conditions and strong winds. The key underpinning threat to this ecosystem type is climate change, which reduces periglacial activity, promotes competitive vegetative invasions of tall shrubs and grasses, and increases fire severity and frequency (which is a threatening process in NSW only). This ecosystem type is also highly sensitive to trampling, as it is unable to regenerate on frequently hiked trails.

The status of the ecosystem type is **Vulnerable** due to its small Extent of Occurrence and the continuing declines from the threat posed by climate change (sub-criterion B1) (Table 23).

Table 23. Summary of the IUCN Red List of Ecosystems assessment of the Australian *Fjaeldmark*.

Criteria	**A**	**B**	**C**	**D**	**E**	**Overall**
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	NT	LC	DD	DD	VU
Sub-criterion 2 *A, C, D: 50-year period including present & future* *B: AOO*	DD	VU	LC	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	LC	LC	LC	DD		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria

Ecosystem Description

Ecosystem Classification

In the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022a), *Fjaeldmark* (also known as Feldmark) is a sub-global ecosystem type (Level 6) belonging to Ecosystem Functional Group *T6.4 Temperate alpine grasslands and* shrublands, within the *T6. Polar/alpine (cryogenic)* biome.

Australian *Fjaeldmark* is equivalent to *Feldmark/Fjaeldmark* in Venn et al. (2017), *Epacris microphylla – Chionohebe pulvinatus* (Group X, Community 40) in Mcdougall & Walsh (2007), *Feldmark Epacris – Chionohebe* or *Veronica* in Costin (1954) and Costin et al. (2000), *Fjaeldmark* in Kirkpatrick (1997) and *Epacris 'microphylla' – Veronica densifolia* association in McVean (1969). However, this ecosystem type is collectively defined by its low vegetation cover that forms on fissile sedimentary rock of exposed ridgelines and summits. It occurs within the Australian Alps Bioregion, Tasmanian Southern Ranges, Tasmanian Central Highlands and Tasmanian West bioregions (SEWPaC 2012; Kitchener & Harris 2013) where fissile sedimentary rock coincides with wind-exposed ridgelines, cols and summits.

Fjaeldmark is not listed under the *Environment Protection and Biodiversity Conservation (EPBC) Act 1999* (EPBC Act). However, *Fjaeldmark* occurs within National Heritage areas and is therefore considered as a 'Matter of National Environmental Significance' under the EPBC Act. In New South Wales, the ecosystem type (classed as 'windswept *Fjaeldmark*') is listed as a Critically Endangered Ecological Community under the Threatened Species Conservation Act 1995 (listed on 2 October 2015) (NSW Threatened Species Scientific Committee 2018).

Distinction from similar ecosystem types

Fjaeldmark can be distinguished from *Alpine-subalpine Open Grassy Heathland and Herbfield*, *Alpine-subalpine Closed Heath* and *Tasmanian Alpine Heath* by the limited vegetation cover (< 50%, with flora short (< 0.4 m) or prostrate in form), low beta diversity, and high cover of pavements of fractured rocks. This ecosystem type is also limited to alpine ridgelines and summits and is only found above the tree line. In contrast, *Alpine-subalpine Open Grassy Heathland and Herbfield* occurs on shallow slopes and summits across the Australian mainland and Tasmania. It is characterised by forbs, grasses, and only a short, often prostrate dwarf shrub canopy (0.1-0.5 m; 0-30% cover) of predominantly non-resprouting (obligate seeding) shrub species (Williams et al. 1988). *Alpine-subalpine Closed Heath* occurs on steep and exposed landscapes of mainland Australia, usually at lower elevations, and is characterised by a dense, tall, shrub canopy (< 4.0 m, ≥ 70% coverage) of both post-fire resprouting and obligate seeding shrub species, with a limited understory of grasses, forbs, and herbs (Williams & Ashton 1988). *Tasmanian Alpine Heath* is dominated by scleromorphic shrubs that form a closed to relatively open canopy and forms on a range of substrates, from fertile to infertile soils, rocky to shallow mineral soils, and poorly drained to well-drained soils (Kirkpatrick et al. 2002).

The ecosystem type often forms mosaics with other similar ecosystem types (Kirkpatrick et al. 2002). The high degree of exposed rock makes boundaries between *Fjaeldmark* easier to differentiate compared to other ecosystem types listed above. A related ecosystem type to *Fjaeldmark* occurs in the subantarctic islands (e.g., Heard and Macquarie islands). This was not included in this assessment.

Distribution

Alpine and subalpine ecosystems occupy ca. 0.15% of Australia (Costin et al. 2000). *Fjaeldmark* is one of the most restricted of these ecosystems (Figure 37), covering < 0.05% of the alpine-subalpine area (Costin 1954; Kitchener & Harris 2013; Mackey et al. 2015). *Fjaeldmark* occurs across a c. 6° latitudinal range in the high mountain environments of south-eastern Australia (New South Wales and Tasmanian; Figure 37), between 145.61° to 148.33° longitude and -36.40° and -43.50° latitude. This ecosystem type occurs from approximately 2000 m to 2150 m above sea level (ASL) in New South Wales (Costin 1954; Costin et al. 2000; McDougall & Walsh 2007), and > 900 m ASL. in Tasmania (Kitchener & Harris 2013). This ecosystem type is restricted to exposed high alpine ridgelines and summits along the Main Range of the Snowy Mountains, New South Wales, mostly between Mount Twynam and Rawson Pass (McDougall & Walsh 2007), and throughout the exposed ridgelines and summits of the central highlands and western and southern mountains in Tasmania (Kitchener & Harris 2013; Annandale & Kirkpatrick 2017).

Fjaeldmark is restricted to between 3.24 km^2 and 4.06 km^2 (CSIRO 1972; Tasmanian Goverment 2014), with present-day mapping produced for this assessment indicating it occupies 3.26 km^2. Spatial products used to create the present-day distribution map represent the most extensive and/or accurate mapping available within each state for the defined bounds of *Fjaeldmark* at the time of assessment (April 2021), as advised by representatives from each government department involved in the assessment process (see main methods).

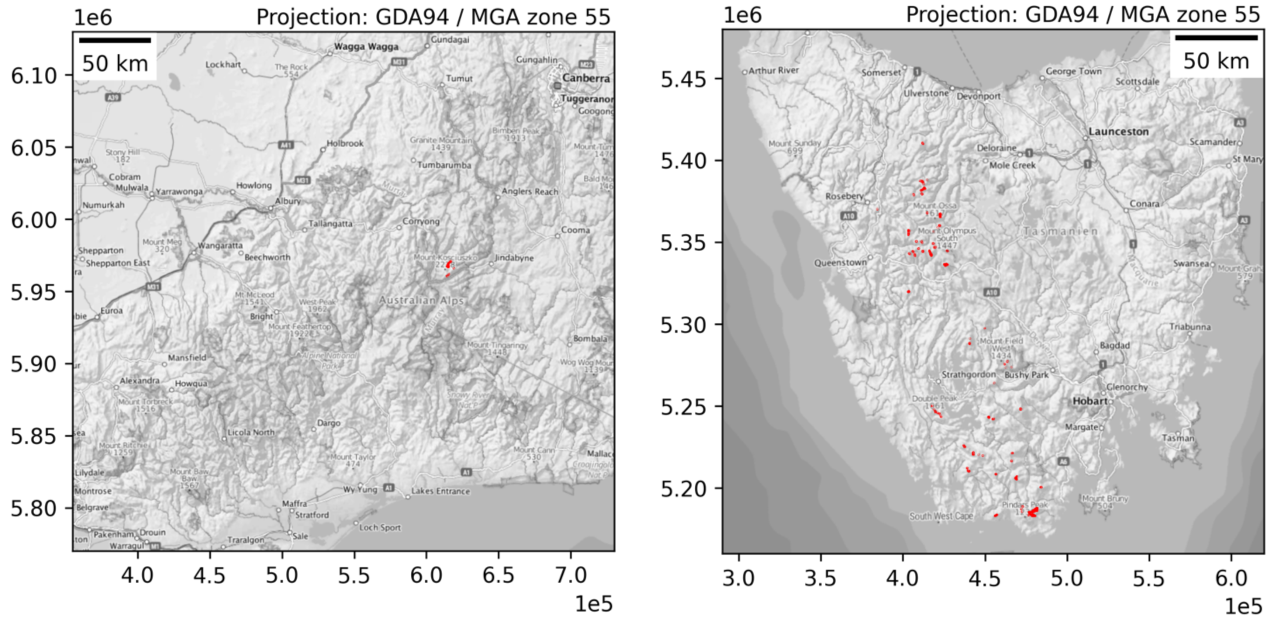


Figure 37. Distribution of *Fjaeldmar*k (red) across the Australian mainland (left) and Tasmania (right).

Abiotic environment

The fundamental abiotic factor that determines the distribution of *Fjaeldmark* is topographical exposure to prevailing winds. This ecosystem type is limited to exposed alpine and subalpine ridgelines and summits that experience relatively low annual mean temperatures (< 8°C), high annual precipitation (> 2500 mm), needle ice formation, periodic desiccation, and strong, prevailing winds (Costin 1954, 1967; Annandale & Kirkpatrick 2017; Venn et al. 2017). In New South Wales, the minimum temperature is below freezing for 6 to 8 months each year and there are approximately 200 freeze cycles per annum (Costin 1954; Barrow et al. 1968; Costin et al. 2000). Most of the precipitation (~60%) falls as snow on the mainland and as rain in Tasmania (Nunez et al. 1996; Costin et al. 2000). Insulating snow cover is limited on the exposed ridgelines and summits where *Fjaeldmark* occurs, exposing the biota to ice-particle abrasion, frost heaving and needle ice formation on bare ground (Barrow et al. 1968; Annandale & Kirkpatrick 2017), as the minimum ground temperatures are considerably lower than in more protected localities (Costin 1954). With the rapid removal of snow by strong winds, alternate freezing and thawing of soils considerably impacts seedling growth and establishment (Costin 1954).

Another key abiotic determinant of this ecosystem type is periglacial solifluction, the gradual mass movement downslope due to freeze-thaw activity. In *Fjaeldmark*, this process causes terraces or stripes, often covered by pavements of fractured rocks. The risers, which capture the sediment dislodged by needle ice, are steeper and well-vegetated (Figure 38) (Costin 1954; Kirkpatrick & Harwood 1980; Kirkpatrick 1984a; McDougall & Walsh 2007). In New South Wales, this rocky substrate is composed of wind-eroded metasediments, lithosols and loose sedimentary shale (Costin 1954; McDougall & Walsh 2007; Mackey et al. 2015), and fissile mudstones in Tasmania (Kirkpatrick 1997; Forsyth 2003). These stripes and terraces are caused by frost creep (downslope displacement of soil during freeze-thaw cycles) from needle ice (needle-shaped ice columns formed when soil temperature is < 0°C and air temperature is > 0°C) which is captured by vegetation (Costin 1967; Barrow et al. 1968; Slee et al. 2016). Soils are shallow, with coarse rocks and exhibit low porosity, low water holding capacity (Treby et al. 2024). Nitrogen values among soils are much the same (0.3%), with a tendency for slightly higher values in the soils under vegetation, which are also slightly less acidic (Barrow et al. 1968).

Fires are an infrequent occurrence in this ecosystem type. Fire intervals throughout the Australian alps region prior to colonisation are estimated between 90 and 120 years on the Australian mainland (Zylstra 2006); the sparse nature and low connectivity of this ecosystem type suggest that historic fire intervals in *Fjaeldmark* may be longer.

Figure 38. *Fjaeldmark* in non-sorted steppe solifluction terrace on Mount Northcote, Kosciuszko National Park, NSW, at elevation of ca. 2100 m, in February 2020. Flora shown here consists of prostrate shrubs and herbs. Downslope heathland and grassland communities forming on deeper soil profiles adjacent to a lake are shown in the top right of the image. Photo: Brodie Verrall.

Characteristic native biota

Unlike nearly all other alpine and subalpine ecosystem types, *Fjaeldmark* often has limited vegetation cover (c. 50%) and low beta diversity (Figure 39, 40). There is a high proportion of chamaephytes and hemicryptophytes, including scattered dwarf prostrate plants, and mat or cushion plants (Kirkpatrick 1997; Costin et al. 2000; McDougall & Walsh 2007). Due to the harsh environmental conditions, the dominant growth forms have morphological and physiological adaptations to low temperatures and severe wind-pruning for long periods (i.e., acaulescent forbs, cushion plants, layering shrubs and bryophytes) (Venn et al. 2017). Vegetation growth rates are characteristically slow and seed development following pollination may take several years. Dominant chamaephyte shrubs are pruned on the windward side and sprout on the sheltered side (Barrow et al. 1968; Annandale & Kirkpatrick 2017) and are considered facilitative nurse plants that are important drivers of community structure (Ballantyne & Pickering 2015b; Verrall 2018).

In New South Wales, *Fjaeldmark* is comprised of a distinct assemblage of 36 taxa where diagnostic species include *Epacris microphylla* s.l., *Colobanthus pulvinatus*, *Ewartia nubigena*, *Luzula australasica* subsp. *dura* and *Poa fawcettiae*. Less common species include *Trisetum spicatum*, *Agrostis muelleriana*, *Leucochrysum alpinum*, *Senecio pectinatus* var. *major*, *Celmisia costiniana* and *Celmisia pugioniformis* but most plants recorded in *Fjaeldmark* also occur in other ecosystem types (McVean 1969; Costin et al. 2000; McDougall & Walsh 2007). The ecosystem type is distinguished by cushion plants, prostrate forbs and crustose lichens that are largely restricted to *Fjaeldmark* (e.g., *Colobanthus pulvinatus*, *Veronica densifolia* and *Kelleria dieffenbachia*) (Costin

1954; McVean 1969). Several other species are also largely restricted to this ecosystem type, including the entire Australian population of *Rytidosperma pumilum*, and most of the distribution of two endemic forbs, *Rannunculus acrophilus* and *Eurphrasia collina* subsp. *lapidosa* (Costin et al. 2000; McDougall & Walsh 2007). There is a gradual transition from *Fjaeldmark* to *Alpine-subalpine Closed Heath* dominated by *Epacris*, but any attempt to unite the floral species lists results in a heterogenous association (McVean 1969; McDougall & Walsh 2007), justifying an arbitrary division.

Conversely, species assemblages in Tasmania are highly variable, lack marker species and are similar to surrounding heathlands (e.g., *Tasmanian Alpine Heath, Coniferous Heath*) (Kirkpatrick 1997; Annandale & Kirkpatrick 2017; Venn et al. 2017). Mat heath is a key component, comprising of scleromorphic shrubs (e.g., *Gaultheria depressa*, *Pentachondra pumila*, *Cryptandra alpina*) forming mats < 5 cm tall (Venn et al. 2017). Alternatively, vegetation may be dominated by cushion plants (such as *Colobanthus pulvinatus*) or shrubs (such as *Orites revoluta*, *Ozothamnus rodwayi*, *Exocarpos humifusus* or *Leptospermum rupestre*) or by bolster plants, particularly *Dracophyllum minimum* (Kirkpatrick 1997).

There is little evidence that faunal assemblages contribute to the identity of *Fjaeldmark* (Costin 1954; Green & Osborne 2012). Subterranean communities appear to be virtually absent, but a few invertebrate seasonal immigrants and alpine specialists have been observed using this ecosystem type (Costin 1954; Williams et al. 2014; pers. comm. K. Green 2019). Most invertebrates captured via pitfall trapping in New South Wales were Dipterans (81%), Hymenopterans (8%) and Collembolans (7%) (Green 1988).

Figure 39. Typical floristic structure of windswept *Fjaeldmark* (*Epacris – Chionohebe*/*Veronica* alliance) on the Main Range, Kosciuszko National Park, NSW, at an elevation of c. 2000 m. Photo: Brodie Verrall (February 2020).

Figure 40. Tasmanian *Fjaeldmark* at The Boomerang (Southern Ranges, Tasmania). Photos: Micah Visoiu (2013).

Key components, processes, and interactions

Wind abrasion and periglacial activity

The principal determinant of *Fjaeldmark* is topographical exposure and extreme abiotic conditions experienced on windswept ridgelines and summits (Figure 41, 42) (Costin 1954; Barrow et al. 1968; Annandale & Kirkpatrick 2017). Severe, unidirectional winds and ice-particle abrasion result in wind-pruned vegetation with the simultaneous erosion on the windward side and renewal on the leeward side (Figure 41, 42) (Barrow et al. 1968). On relatively uneven terrain, where projecting rocks offer local but permanent protection from the prevailing winds, vegetation is relatively stable both in space and time. On more even surfaces, there is obvious erosion of the vegetation and its surface soil, resulting in a slow migration of the ecosystem type in the direction of the prevailing winds (Costin 1954; Barrow et al. 1968; Lynch & Kirkpatrick 1995). These winds also erode soil and gravel that accumulate among the living vegetation, eventually causing their death by smothering and abrasion (Costin 1954). Erosion can also redistribute soil stored seedbank and offer opportunities for colonisation (Lynch & Kirkpatrick 1995; Kirkpatrick et al. 2002). Severe winds also displace insulating winter snow cover and expose the soils and biota to freezing temperatures, which promotes periglacial solifluction activity (e.g., frost creep, needle ice) (Costin 1954). These periglacial solifluction processes result in the sorting of scree steppes and determines cyclidic dynamics of *Fjaeldmark* vegetation (Barrow et al. 1968; Lynch & Kirkpatrick 1995; Annandale & Kirkpatrick 2017). Subsequently, positive plant interactions have been observed in this ecosystem type, where facilitative nurse shrubs are drivers of ecosystem diversity and floristic assemblage (McDougall & Wright 2004; Ballantyne & Pickering 2015b).

These processes promote and maintain the floristic structure and assemblage of *Fjaeldmark*, but the extreme abiotic conditions restrict most fauna, except for the a few species of invertebrates documented pollinating and grazing on plants in this ecosystem type (Green & Osborne 2012).

Figure 41. *Fjaeldmark* on the Boomerang, Tasmania, with cushion plant showing erosion on the southern side and growth on the northern side in response to southerly winds. North at top. Photo: Micah Visoiu.

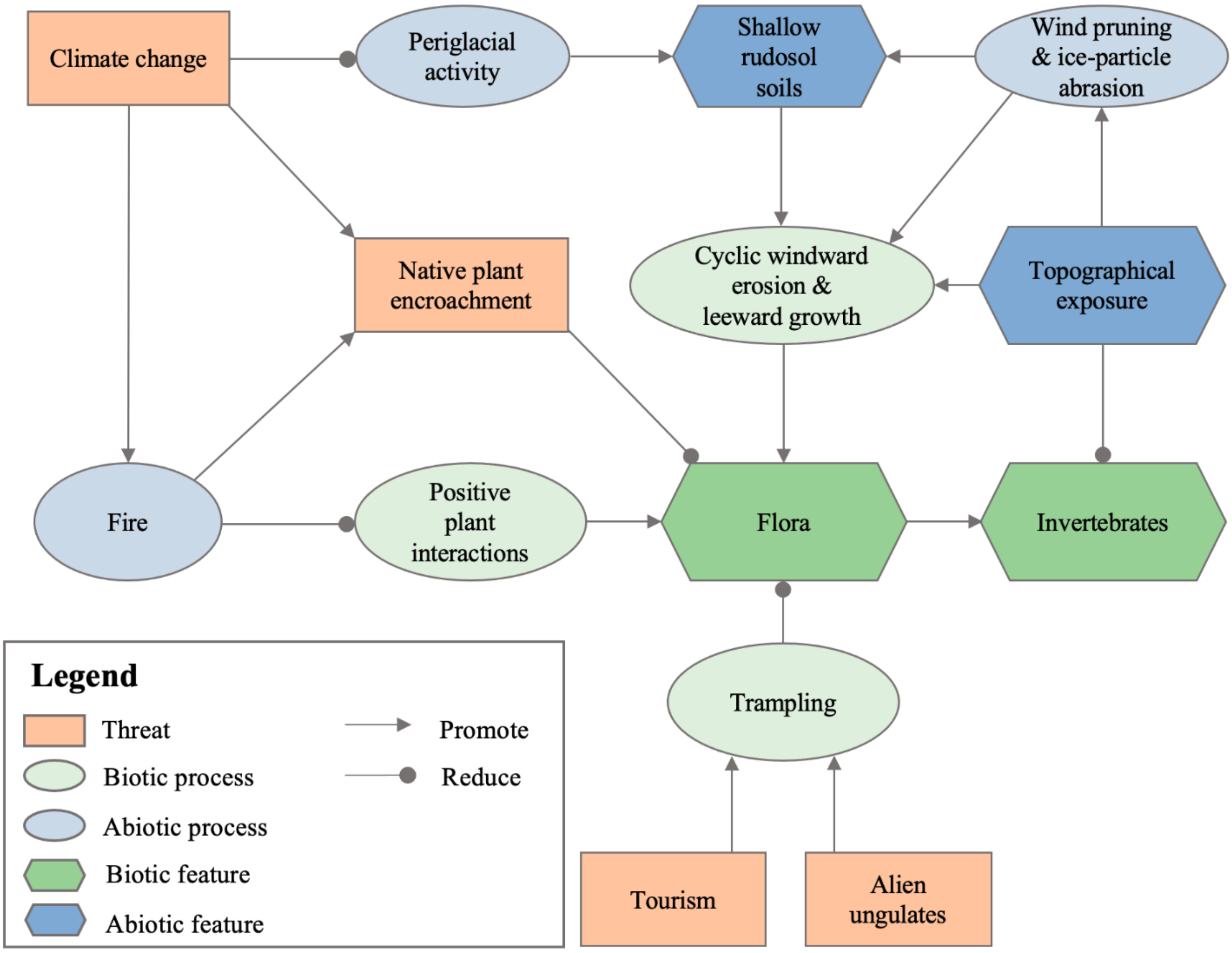


Figure 42. Conceptual model for the Australian *Fjaeldmark.*

Major threats

Climate warming

Climate warming may to lead to encroachment of grass/mat heath from adjacent ecosystem types such as *Alpine-subalpine Closed Heath*, and *Alpine-subalpine Open Grassy Heathland and Herbfield* (Annandale & Kirkpatrick 2017; Verrall 2018; NSW OEH 2019). In the Snowy Mountains of New South Wales, the increase in grassy vegetation and litter produces relatively nutrient-rich soils. This may create more favourable conditions for plants and drive the break-down of the facilitative interaction between the nurse shrubs (*Epacris microphylla* s.l.) and the plants sheltering under their canopy (Verrall 2018). Once vegetation cover increases, it offers additional insulation against needle ice activity by retaining snow cover, which may promote further vegetative invasion and form a positive feedback loop. In Tasmania, climate warming may lead to the encroachment of adjacent shrubs, including *Baeckea gunniana*, and sedges, herbs and rushes. These can grow over the stony surface and insulate the soil, preventing needle ice from forming (Luthin & Guymon 1974). Additional vegetation also traps fine soil material, further reducing the needle-ice formation and frost creep that maintain *Fjaledmark* landforms. Increasing temperatures with climate change are likely to disrupt the periglacial geomorphic processes that maintain this ecosystem type including the solifluction processes that maintain scree steppes (Costin 1954; Barrow et al. 1968; Lynch & Kirkpatrick 1995), with litterfall from the taller vegetation canopy insulating soils and further.

Fires

Climate change is predicted to increase the frequency and intensity of fires in alpine ecosystems (Zylstra 2018). In New South Wales, even low intensity burns have been shown to degrade floral structure and assemblage of *Fjaeldmark* and affects the facilitative capacity of nurse shrubs (Verrall 2018). In the changing climate, a single fire can substantially alter the vegetation community composition through initial reduced shrub cover (*Epacris microphylla* s.l.) and higher graminoid (*Poa fawcettiae*) and herb cover compared to unburnt *Fjaeldmark* (Verrall 2018). The shrubs that grow back may be taller in stature than the original prostrate nurse shrubs, further reducing the relative importance and benefits of the characteristic facilitative interactions (Verrall 2018). Consequently, the ecosystem type may transition into *Alpine-subalpine Open Grassy Heathland and Herbfield* and/or *Alpine-subalpine Closed Heath* on the mainland.

Fire at a frequency that outpaces the regeneration of obligate seeding shrubs is likely to result in competitive vegetation invasions (15 years) (Walsh & Mcdougall 2004; McDougall & Walsh 2007; Venn et al. 2016; Verrall 2018). *Fjaeldmark* is unlikely to burn due to its exposed high-elevation position, low-stature vegetation, and lack of connectivity with more flammable ecosystem types and between constituent flora. However, chamaephyte shrubs are flammable, and the flora of surrounding *Alpine-subalpine Closed Heath* on the Australian mainland is particularly flammable, increasing its propensity to burn (and thus chance of ignition into adjacent *Fjaeldmark*). The ecosystem type's flammability may increase if competitive encroachment/invasion of flammable vegetation such as tall grasses and mat heaths occurs due to climate warming. In contrast, fires in Tasmania have previously maintained *Fjaeldmark* by hindering shrub encroachment (Annandale & Kirkpatrick 2017) and is therefore fire not considered a threat to ecosystem extents in Tasmania.

Trampling and dispersal

Human trampling compacts soils, causes erosion and limits vegetative regeneration (McDougall & Wright 2004; Ballantyne et al. 2014a; Ballantyne & Pickering 2015b). Wind-propelled vegetation is unable to cross hiking trails, which disrupts cyclidic vegetation dynamics that determine the floristic assemblages via degradation and renewal along the wind exposure gradient (McDougall & Wright 2004; Ballantyne et al. 2014a).. Exposed soil can be subsequently colonised by the dominant species of the adjoining *Alpine-subalpine Open Grassy Heathland and Herbfield*, including *Celmisia costiniana* and *Aciphylla glacialis*. In New South Wales, there is growing concern about the trampling impacts from feral ungulates (NSW OEH 2019). Tourists also act as a dispersal vector for invasive flora and disease; *Acetosella vulgaris* and *Hypochaeris glacialis* have been documented in mainland extents of this ecosystem type (McDougall & Walsh 2007).

IUCN Stresses Classification

1.2 – Ecosystem degradation (Altered temperature/precipitation/fire regime)
2.3.2 – Indirect species effects (Competition)

IUCN Threats Classification

6.1 – Recreational Activities (hikers, any others)
7.1.1 – Increase in fire frequency/intensity (NSW only)
8.1.2 – Invasive non-native species (fauna: feral horses, sambar deer, fallow deer; flora: *Acetosella vulgaris*, *Hypochaeris glacialis*)
8.2.1. – Problematic native species (unspecified species, tall shrubs, grasses and herbs)
11.3 – Temperature extremes

Ecosystem collapse

Fjaeldmark can collapse and transition into *Alpine-subalpine Open Grassy Heathland and Herbfield* or *Alpine-subalpine Closed Heath*. This may occur due to increases in cover of tall shrubs and grasses and losses in the characteristic facilitative interactions between low-lying nurse shrubs and underlying vegetation. Under a warmer climate, vegetation may survive beyond the shrub canopy, increasing the vegetation cover and thus promoting accumulation of soils and snowpack. In NSW only, increases in the occurrence of fires may accelerate the shift to taller vegetation (Pickering & Venn 2013; Verrall 2018; Zylstra 2018) that is occurring due to the warming climate (J. Kirkpatrick, pers. comm.); fires are likely to moderate shrub growth and encroachment of ecosystem extents in Tasmania (Kirkpatrick & Bridle 2013).

In this assessment, *Fjaeldmark* is considered collapsed when any of the following occurs:

1. *Area*: The mapped distribution declines to zero (100% loss) (criteria A and B), or;
2. *Fire frequency*: The frequency of fires increases to 1 every 15 years (criterion C; NSW only), and/or;
3. *Vegetation cover:* The cover of the vegetation, predominantly shrub and tall grasses, is ≥ 70–75% of the ecosystem area (criterion D).

Risk Assessment

Criterion A: Reduction in distribution

Summary

The risk status of *Fjaeldmark* is **Least Concern** under sub-criterion A1, **Data Deficient** under sub-criterion A2. and **Least Concern** under sub-criterion A3.

Methods

Time-series maps of *Fjaeldmark* were unavailable. We based the assessment of sub-criterion A1 on changes in *Fjaeldmark* area based on recent (2013) (Annandale & Kirkpatrick 2017) and historical aerial photographs (26-59 years prior)(DPIPWE 2016a) and data from 38 *Fjaeldmark* patches across Tasmania. Data to predict future changes in the distribution of *Fjaeldmark* under sub-criterion A2 were unavailable. We inferred historical change for sub-criterion A3 based on accounts of change in land use practices or invasion of introduced species (Costin 1954).

Assessment outcome

Recent change (A1)

Data were unavailable to estimate the change in area over the past 50 years in *Fjaeldmark* in New South Wales. Annandale & Kirkpatrick (2017) collated data on change in area of 38 *Fjaeldmark* patches in Tasmania based on recent (2013) and historical aerial photographs from 26-59 years prior (Figure 43) (DPIPWE 2016a). These data reveal the area of *Fjaeldmark* in Tasmania has on average slightly but significantly increased over time ($t_{37} = 3.591$, $P = 0.001$). As the Tasmania distribution accounts for over 90% of the Australian distribution, the risk status is **Least Concern** under sub-criterion A1.

Future change (A2)

Based solely on the trend in area of *Fjaeldmark* in Tasmania (see sub-criterion A1), *Fjaeldmark* may continue to increase in area. This is likely because Tasmania is colder and drier than the mainland and the cold temperatures and insulating snow may buffer against shrub thickening and encroachment. However, future declines in *Fjaeldmark* throughout mainland extents are likely due to the indirect effects of climate change. Warming may reduce the characteristic periglacial activity that maintains the structure and low cover of characteristic biota (e.g., needle ice, frost creep, and solifluction) (Costin 1954; Barrow et al. 1968; Lynch & Kirkpatrick 1995). Increasing ambient temperatures will improve both the growing conditions and thermal energy available to support taller, dense flora growth and encroachment from adjacent ecosystem types (Pickering & Armstrong 2014; Annandale & Kirkpatrick 2017) or thickening and growth of in-situ shrubs may drive ecosystem declines. Climate change is also likely to alter fire regimes by increasing the frequency and severity of fires in *Fjaeldmark* leading to the loss of characteristic biota (short fire intervals) and creation of bare ground for recruitment of non-characteristic biota (Pickering and Venn, 2013; Verrall, 2018; Zylstra, 2018). In the current warming climate, fire can cause the rapid expansion of vegetation cover from adjacent ecosystem type, such as *Alpine-subalpine Closed Heath* (Verrall 2018). However, we lack sufficient data to quantitative project declines in area due to encroachment. The risk status is **Data Deficient** under sub-criterion A2.

Historical change (A3)

Fjaeldmark ecosystem types are not important economically. Unlike adjacent ecosystem types, *Fjaeldmark* lacks the soils necessary to support the development of palatable pastures for grazing, and thus have were not substantially affected by changes in land use practices or grazing livestock (Costin 1954). Localised declines have also been documented due to trampling by tourists in the summer seasons (Ballantyne et al. 2014b; Ballantyne & Pickering 2015a). Therefore, we assumed that it is unlikely for change in area to meet the threshold for Vulnerable (> 50% loss). The risk status is **Least Concern** under sub-criterion A3.

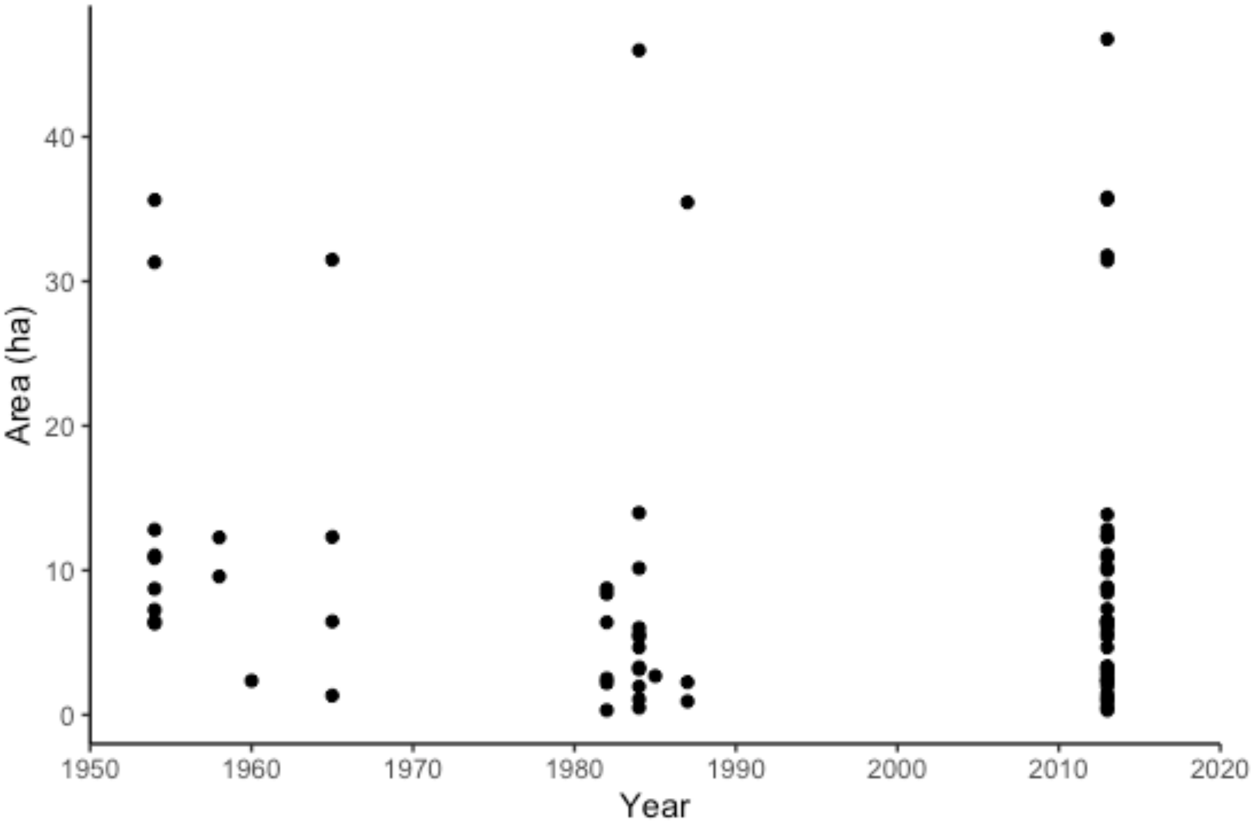


Figure 43. Area of 33 *Fjaeldmark* patches in Tasmania based on aerial photographs (Annandale & Kirkpatrick 2017)

Criterion B: Restricted distribution

Summary

The risk status of *Fjaeldmark is* **Near Threatened** under sub-criterion B1, **Vulnerable** under sub-criterion B2, and **Least Concern** under sub-criterion B3.

Methods

Under criterion B, the extent of occurrence (EOO; sub-criterion B1) and area of occupancy (AOO; sub-criterion B2) were determined using a combination of existing maps from New South Wales (CSIRO 1972) and Tasmania (Tasmanian Goverment 2014) (Figure 44). The distribution map was created by cropping *Fjaeldmark* to minimum elevation thresholds; ≥ 900 m for Tasmania, and ≥ 2000 m for NSW/ACT. Subsequently, areas of *Fjaeldmark* overlapping with existing distribution maps of *Snowpatch Herbfield* and *Alpine-subalpine Lakes* were removed from the distribution to produce a final 'minimum' extent of *Fjaeldmark*.

The number of locations was based on fire as this is the most important stochastic threat to the ecosystem type, although it only relevant to the mainland (sub-criterion B3).

Assessment outcome

Sub-criterion B1

The Extent of Occurrence (EOO) for Australian *Fjaeldmark* is currently estimated at 55,024.01 km^2, (Figure 44). The EOO nears but does not meet the threshold for Vulnerable, and so the risk status is **Near Threatened** under sub-criterion B1.

Sub-criterion B2

Australian *Fjaeldmark* occurs across 30 10×10 km grid cells (AOO). There is evidence that the ecosystem type may be at risk from changes in the vegetation assemblage and structure due to the predicted increased in degree days and thus longer growing seasons, which support the invasion of the bare areas that characterise the community by vegetation (see criteria C below). The risk status of the ecosystem type is therefore **Vulnerable** under sub-criterion B2.

Sub-criterion B3

The major stochastic threat to *Fjaeldmark* is fire occurring at intervals of < 15 years (Venn et al. 2016; Verrall 2018). While fire is currently an uncommon event in the Australian alpine and subalpine areas (Zylstra 2006, 2018; Williams et al. 2008), the frequency of fires is projected to increase (see criterion C). However, fire only poses a threat to the NSW distribution of the ecosystem type, which represents < 10% of the whole distribution. Therefore, the risk status of the ecosystem type is **Least Concern** under sub-criterion B3.

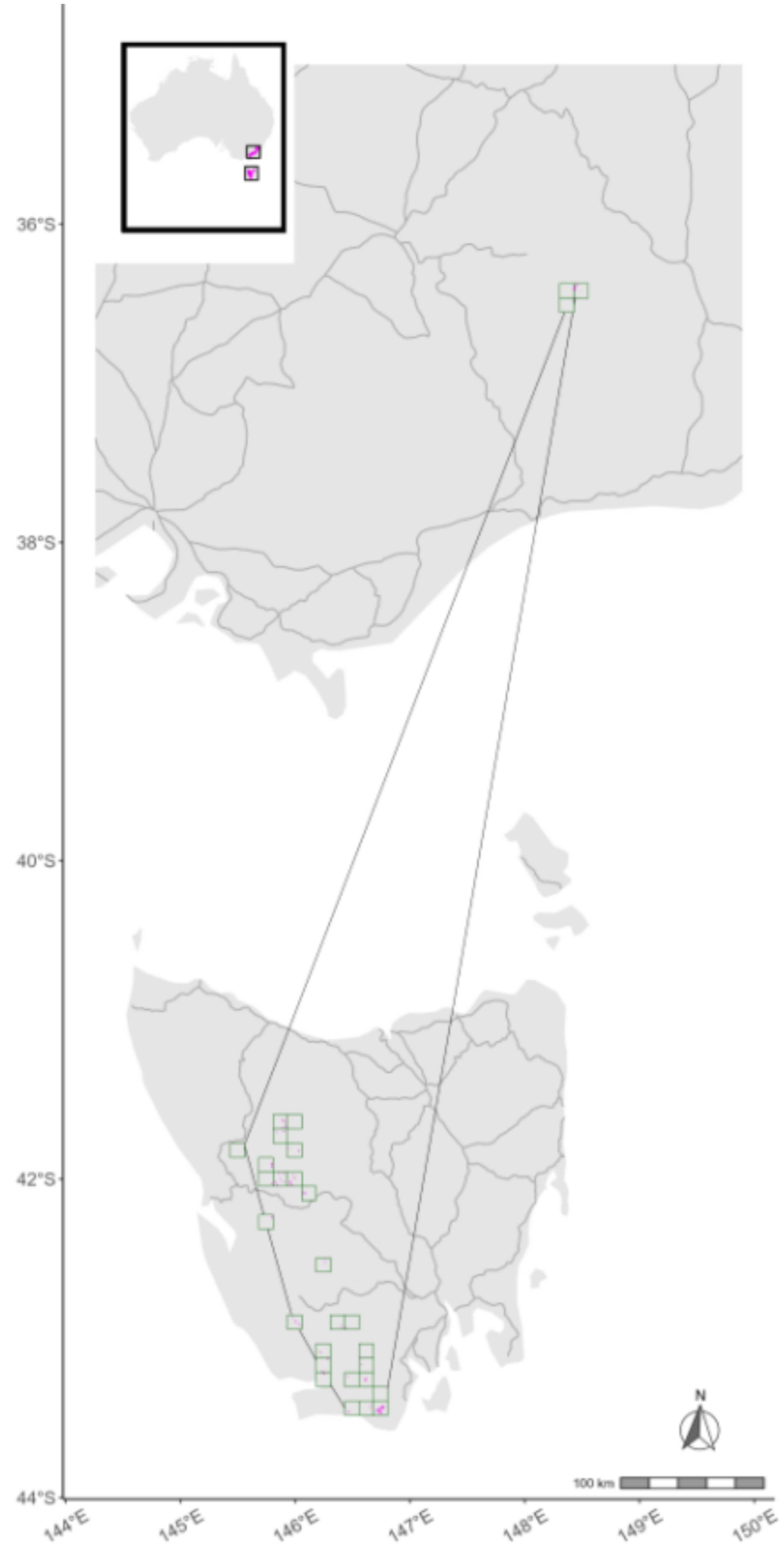


Figure 44. Map of *Fjaeldmark* (magenta polygons) showing EOO (black polygon) and AOO (green squares.

Criterion C: Environmental degradation

Summary

The risk status of *Fjaeldmark* is **Least Concern** under sub-criteria C1, C2, and C3.

Identification of abiotic indicator

We examined the relevance and data availability/quality for four potential indicators to assess the risk of collapse from environmental degradation:

- *Fire frequency*: a direct measure of the frequency of fires.
- *Degree days*: a measure of temperature and used to estimate changes in the length of growing season which facilitates plant invasions.
- *Mean daily minimum temperature in the coldest month:* a measure of temperature used to estimate changes in periglacial activity.

Indicator: Fire frequency

Relevance to ecosystem function

Fires are a rare event in *Fjaeldmark* due to its high elevation, low-statured plants and lack of connectivity. Historically, fire may have had a positive impact in the ecosystem type as it can create bare ground and hinder shrub encroachment (Annandale & Kirkpatrick 2017). In Tasmania, the areas of *Fjaeldmark* that burnt in 1934 showed a decrease in vegetation cover, which gradually recovered over decades (Annandale & Kirkpatrick 2017). However, climate change will likely increase the frequency and severity of fires in *Fjaeldmark*; the creation of bare ground following fire may enable recruitment of flora from other ecosystem types (Hickman et al. 2024) and thus fire is a threatening process to extents in NSW (Pickering & Venn 2013; Verrall 2018; Zylstra 2018). The altered fire regimes in NSW will likely interact with the warming alpine climate to accelerate the shift in the vegetation community (J. Kirkpatrick, pers. comm.) through initial reduced shrub cover (*Epacris microphylla* s.l.) and higher graminoid (*Poa fawcettiae*) and herb cover compared to unburnt *Fjaeldmark* (Verrall 2018). The shrubs that grow back may be taller in stature than the original prostrate nurse shrubs, further reducing the relative importance and benefits of the characteristic facilitative interactions (Verrall 2018). Consequently, the ecosystem type may transition into *Alpine-subalpine Open Grassy Heathland and Herbfield* and/or *Alpine-subalpine Closed Heath* on the mainland.

Selection of collapse threshold

Fire kills *Epacris microphylla* s.l. and the shrubs regenerate from the soil seedbank (obligate seeders) (Walsh & Mcdougall 2004). Recovery of *Epacris microphylla* s.l. after fire is slow. Shrub cover was < 1% nine years post-fire and increased to 20% 15 years post-fire, half that of the coverage in unburnt sites (40%) (Verrall 2018). These shrubs take five years to mature and produce seeds (Keith 1996). Minimum intervals for fires can be defined based on the time taken for the characteristic nurse shrubs to regenerate and mature to produce seed (Barrett & Yates 2015). Therefore, we consider the ecosystem type to collapse if fire occurs at a frequency of more than 1 in 15 years.

Data availability and quality

Recent change (C1)

The only fire recorded to burn *Fjaeldmark* in New South Wales was in 2003 (Venn et al. 2016). Recovery of *Fjaeldmark* vegetation after the 2003 fire was monitored between 2003 and 2018 (Pickering & Venn 2013; Venn et al. 2016; Verrall 2018). In Tasmania, no fires have burned the *Fjaeldmark* since the 1930s (Annandale & Kirkpatrick 2017).

Future change (C2)

There are insufficient data to reliably estimate future change in occurrence in *Fjaeldmark* in New South Wales. Fire in not considered a threat Tasmania, and therefore this part of the ecosystem type's distribution is not included in this analysis.

Historical change (C3)

Zylstra (2006) outlined the fire history in the alps on mainland Australia. Previous estimates of the frequency of fires in the alpine are 1 in 100 years (Worboys 2003; Pickering & Venn 2013). In New South Wales, the only recorded fire to burn *Fjaeldmark* was in 2003 (Verrall 2018). In Tasmania, the latest fire recorded to burn *Fjaeldmark* in the past 200 years was likely in 1934 at Rocky Hill, while the *Fjaeldmark* around Barn Bluff and Cradle Cirque burned in the late 1800s or early 1900s (Annandale & Kirkpatrick 2017).

Selection of initial and present/future values

Recent change (C1)

In New South Wales, the 2003 fire is the only time that *Fjaeldmark* has been recorded to burn, with 1 ha of *Fjaeldmark* burnt (Verrall 2018). However, previous estimates of the frequency of alpine fires are 1 in 100 years (Worboys 2003; Zylstra 2006; Williams et al. 2008; Pickering & Venn 2013). To define the initial value, we assumed that the initial value in New South Wales may range from zero (as there have been no recorded burns in *Fjaeldmark* before 2003) to 1 in 100, based on the background rate of alpine fires. We estimated that the current value in New South Wales is 1 in 50, as only one fire has occurred in the past 50 years. There have been no fires recorded in *Fjaeldmark* in Tasmania over the past 50 years (Kirkpatrick & Harwood 1980).

Future change (C2)

Fire weather is likely to increase in the future (Clarke et al. 2011). However, we were unable to reliably estimate future change in the frequency of fires in *Fjaeldmark* in New South Wales due to there being too few instances of past fires to reliably project forward.

Historical change (C3)

Annandale & Kirkpatrick (2017) report the fire events in the *Fjaeldmark* in Tasmania. Only one fire is recorded to have burnt the *Fjaeldmark* in New South Wales (Verrall 2018). However, previous estimates of the frequency of alpine fires are 1 in 100 years (Worboys 2003; Zylstra 2006; Williams et al. 2008; Pickering & Venn 2013).

Calculation of severity and extent

Recent change (C1)

Due to the topography, fires in the alpine zone are rare as it required very specific type of weather for *Fjaeldmark* to burn, i.e., warm, dry (Udy et al. 2004) and with sufficient fuel (Fraser et al.

2016). There have been no fires over the past 50 years that have burned *Fjaeldmark* in Tasmania, which accounts for > 90% of the Australian distribution.

In New South Wales, *Fjaeldmark* had not been recorded to burn until the 2003 fires. In 2003, 1 ha was burnt, which accounts for < 1% of the whole ecosystem type. We estimated that the initial fire frequency was zero to 1 in 100 years, and the current frequency is 1 in 50 years. The relative severity of change in fire frequency in New South Wales (7% of the distribution) may range from 100 ´ ((1/100) – (1/50)) / ((1/100) – (1/15)) = 17.6% to 100 ´ (0 – (1/50)) / (0 – (1/15)) = 30%. Based on the combined changes across the ecosystem type the risk status is **Least Concern** under sub-criterion C1. However, these analyses show that extent in New South Wales may be more threatened than the Tasmanian extent if these ecosystem types were assessed at the state level.

Future change (C2)

Fires are predicted to increase in frequency in the future (Clarke et al. 2011). As *Fjaeldmark* in NSW is highly sensitive to burning (see Criterion D), any future fires are likely to cause substantial degradation and local collapse (Verrall 2018, 2023). The predicted increase in fires in adjacent alpine ecosystem types such as *Alpine-subalpine Closed Heath* and *Alpine-subalpine Open Grassy Heathland and Herbfield* (see other assessments in this report) are likely to also mean *Fjaeldmark* may burn at a higher rate than previously. However, we were unable to reliably estimate the future likelihood of *Fjaeldmark* being burnt due to a lack of clear relationship between past increases in fire frequency in the alpine zone and *Fjaeldmark* burning. Given the Tasmanian distribution of the ecosystem type, which accounts for 93% of the ecosystem's distribution, is not considered to be threatened by fire, the risk status of the ecosystem type under sub-criterion C2 (next 50 years) was **Least Concern**.

Historic change (C3)

Subalpine, alpine and high frost hollows were likely not burnt during Indigenous fire management in the Australian Alps (Zylstra 2006; Kirkpatrick & Bridle 2013). Since the ban on grazing and burning for pasture management in 1944, there have been no widespread fires in the Kosciuszko alpine zone except in 2003 (Worboys 2003). However, despite the increased fire frequency, *Fjaeldmark* has rarely burned. The largest fires recorded in the Australian Alps were in 1938/1939 and 2002/2003 where fires spread across most of the subalpine and alpine areas (Zylstra 2006), but *Fjaeldmark* in New South Wales was only recorded to burn in 2003.

In Tasmania, the fire frequency in Tasmanian *Fjaeldmark* ranges between 0 to 2 in 200 years (J. Kirkpatrick, pers. comm.) with most fires in alpine extents a result of lowland fires burning upwards (Kirkpatrick & Bridle 2013). For example, *Fjaeldmark* at Rocky Hill burned in 1934 and may have previously burned within the last one or two centuries (Annandale & Kirkpatrick 2017). Areas in Barn Bluff and Cradle Cirque burned in the late 1800s or early 1900s (Annandale & Kirkpatrick 2017). As there is no evidence that the frequency of fire in *Fjaeldmark* has changed substantially since c. 1750, and it is unlikely to meet the threshold for Vulnerable, the risk status is **Least Concern** under sub-criterion C3.

Indicator: Growing degree days

Relevance to ecosystem function

Periglacial activity is a defining process in *Fjaeldmark*. Freeze-thaw cycles cause the gradual movement of soils down-slope (i.e., periglacial solifluction) and formation of needle ice to create terraces of shallow, rudosol soils covered by fractured scree (Costin 1954; McDougall & Walsh

2007). This process is further promoted by severe winds that remove winter snow cover to expose soils and biota to freezing temperatures. These processes cause the sorting of scree slopes and drive cyclic dynamics of characteristic vegetation, such as the facilitative role of nurse shrubs (Barrow et al. 1968; Lynch & Kirkpatrick 1995; Annandale & Kirkpatrick 2017). However, the warming temperatures under climate change are likely to reduce the periglacial activity, which may promote invasion by competitive plant species as the growing season extends and increases fire severity and frequency (Pickering & Armstrong 2014; Annandale & Kirkpatrick 2017).

Data availability and quality

Predicting future changes in degree days and comparing this to the thermal tolerances of several primary competitive lower-altitude species may provide information on potential increased invasion by non-characteristic plant species under a warming climate. However, insufficient data are available to reliably relate changes in growing degree days to the increased cover of particular species and therefore ecosystem collapse.

Indicator: Mean daily minimum temperature of the coldest month

Relevance to ecosystem function

Measuring soil temperatures can provide information on possible changes in periglacial activity (see Indicator: Growing degree days for further description).

Data availability and quality

Data on soil temperatures across the ecosystem type are unavailable. However, minimum daily temperatures are also linked with the formation of needle ice (Costin 1967; Barrow et al. 1968; Slee et al. 2016). Therefore, measuring the mean daily minimum temperatures of the coldest month can provide information on changes in periglacial activity that support the unique biotic interactions in *Fjaeldmark*. However, insufficient data are available to reliable relate changes in temperature to shifts in vegetation and therefore ecosystem collapse.

Criterion D: Disruption of biotic processes and/or interactions

Summary

The risk status of *Fjaeldmark* is **Data Deficient** under all sub-criteria under criterion D.

Identification of biotic indicator

We examined the relevance and data availability/quality for one indicator to assess the risk of collapse from disruption of biotic processes and/or interactions:

- *Vegetation cover:* a direct measure of the proportion of ground cover of vegetation.

Indicator: Vegetation cover

Relevance to ecosystem function

Fjaeldmark vegetation is characterised by short prostrate nurse shrubs, grasses and herbs (McDougall & Walsh 2007). An increase in tall shrubs and grasses from adjacent *Alpine-subalpine Open Grassy Heathland and Herbfield* would alter the structure of *Fjaeldmark* and cause the loss of the characteristic facilitative interactions between nurse shrubs and underlying vegetation. Plants would increasingly survive and grow outside of the shrub canopy, increasing vegetation cover, and consequently would promote the build-up of soils and promote winter snowpack formation. This would ultimately lead to a shift to *Alpine-subalpine Open Grassy Heathland and Herbfield*.

Data availability and quality

Recent change (D1)

Change in vegetation and ground cover in *Fjaeldmark* in Kosciuszko National Park, New South Wales, was measured between 2003 and 2020 (Pickering & Venn 2013; Venn et al. 2016; Verrall 2023). Vegetation cover was compared in three burnt and three unburnt 30×20 m sites. Cover was estimated at each site using 500-point samples within each plot. Vegetation cover was calculated as 100% minus the cover for bare ground, rock/scree, litter and roots. Data indicating vegetation cover in New South Wales in 1969 were unavailable. Annandale & Kirkpatrick (2017) measured vegetation cover in 38 sites of *Fjaeldmark* across Tasmania. Vegetation cover was estimated based on recent (2013) and historical aerial photographs taken 26-59 years prior (Figure 45) (DPIPWE 2016b).

Future change (D2)

There were insufficient data to reliably estimate future change in vegetation cover of *Fjaeldmark* in New South Wales or Tasmania.

Historical change (D3)

There were insufficient data to reliably estimate historical change in vegetation cover of *Fjaeldmark*.

Selection of collapse threshold

Vegetation cover in *Fjaeldmark* can vary from 30-70%, with an average of 50-60% in New South Wales (Verrall 2018), with an average of 50-60% (Kirkpatrick 1997; Verrall 2018). Therefore, *Fjaeldmark* is considered collapsed once vegetation cover is ≥ 70-75% of the ecosystem area.

Selection of initial and present/future values

Recent change (D1)

To measure changes in vegetation cover in New South Wales over the past 50 years (1969-2019), we fit a generalised additive model to the New South Wales vegetation cover data, with year and whether the site was burn or unburnt as explanatory variables. We used the model trends to extrapolate to 2020 (Figure 45). Data are unavailable to assess change in vegetation in Tasmania.

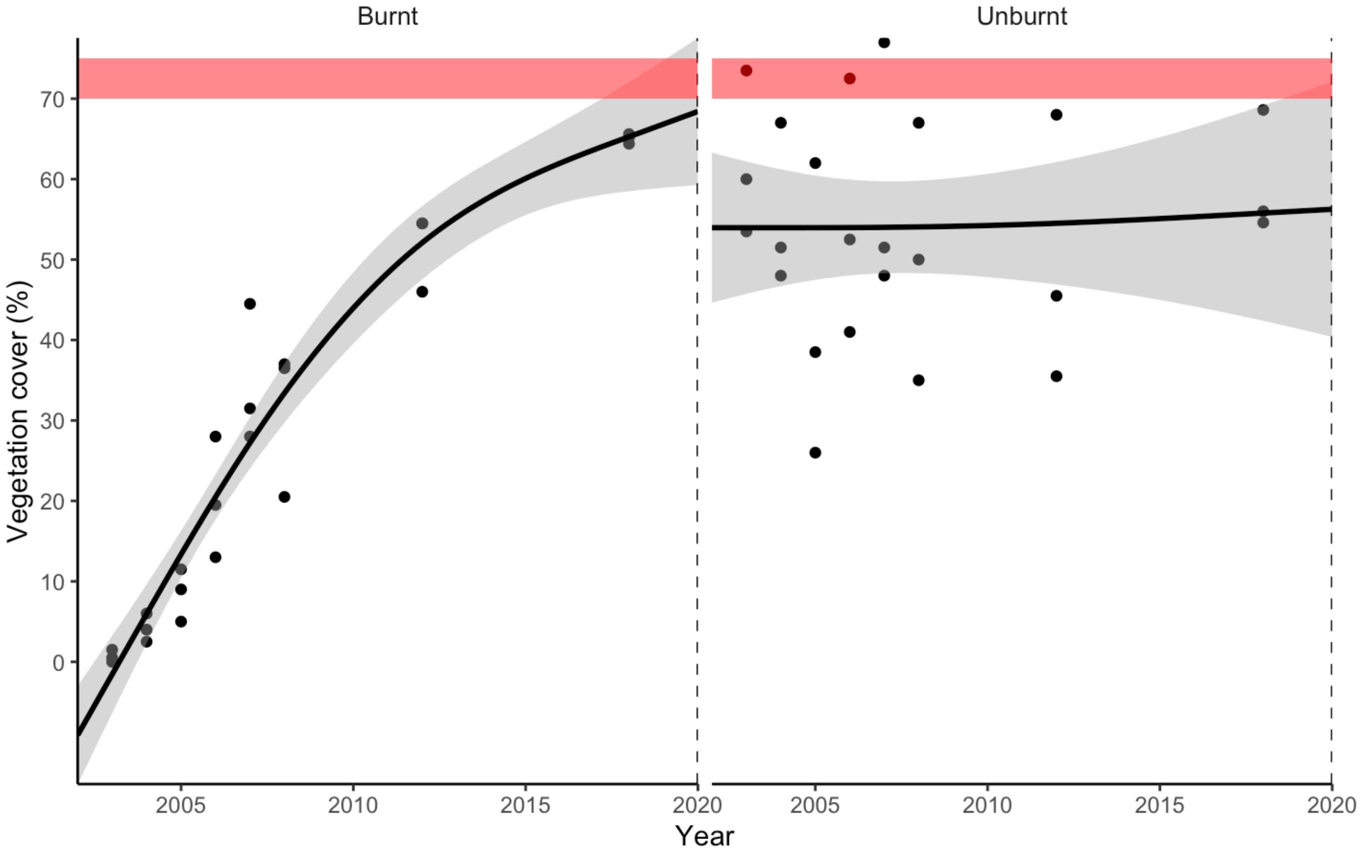


Figure 45. Vegetation cover in burnt and unburnt *Fjaeldmark* in Kosciuszko National Park, New South Wales (Verrall 2018). *Fjaeldmark* sites were burnt in the 2003 fire. Red bar represents the range for the collapse threshold. Vertical dashed line highlights the values in 2020.

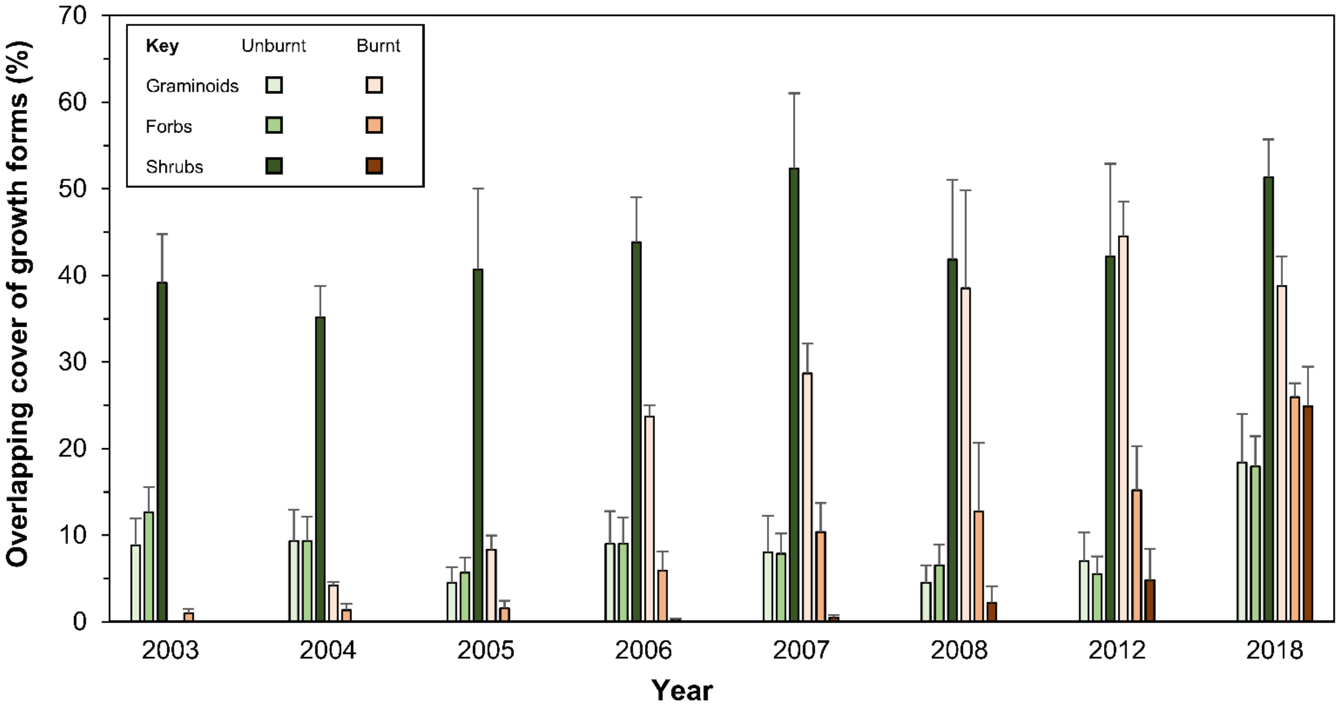


Figure 46. Vegetation cover (%) of three plant growth forms in burnt and unburnt sites of *Fjaeldmark* in Kosciuszko National Park after the fire in 2003 (Source: Verrall 2018).

Future change (D2)

We were unable to reliably estimate the rate of change in vegetation cover of unburnt *Fjaeldmark* in New South Wales due to uncertainty in the modelled change between 2003 and 2018 and lack of data for 1969. Due to insufficient data, we were also unable to reliably estimate change in Tasmania.

Historical change (D3)

We were unable to reliably estimate historical change in vegetation cover of *Fjaeldmark* in New South Wales or Tasmania.

Calculation of severity and extent

Recent change (D1)

In New South Wales, vegetation cover in burnt sites was significantly different to unburnt sites (t_{42}= 9.498, $P < 0.05$) and there was a significant interaction with year (t_{42}= 10.370, $P < 0.05$) (Verrall 2018). In burnt sites, vegetation cover has substantially increased between 2003 and 2018 and was modelled to near the collapse threshold in 2020 (68%; 95% confidence intervals: 59%, 78%), whereas vegetation cover in unburnt sites remained relatively stable (56%; 95% confidence intervals: 40%, 72%). Whilst vegetation cover can recover relatively quickly post-fire, the structure and assemblage of the burnt areas was different compared to the unburnt areas (Figure 46). In burnt areas, the cover of *Poa fawcettiae* was triple that of unburnt areas (Unburnt = 8.79 ± 1.61; Burnt = 25.74 ± 1.18) (Figure 46). The cover of the diagnostic nurse shrub (*Epacris microphylla*) was also half of what is found in unburnt areas (Unburnt = 34.50 ± 3.79; Burnt = 16.56 ± 0.52) (Figure 46).

Unburnt *Fjaeldmark* covered ~1 ha, equating to < 1% of the whole ecosystem distribution. Estimates of vegetation cover in 1969 were unavailable, so we were unable to reliably estimate change in the New South Wales distribution. In Tasmania, vegetation cover has shown a 0.065% average annual increase in vegetation cover, equating to an average increase of 3.25% over 50 years. However, detailed estimates of vegetation cover over the past 50 years were unavailable for Tasmania. The risk status is **Data Deficient** under D1.

Future change (D2)

The data assessed under D1 suggest that the characteristic structure and composition of vegetation in New South Wales may be at risk in the future due to the predicted increase in the frequency of fires in the alps (Clarke et al. 2011). However, we were unable to reliably estimate future change across the ecosystem type. The risk status under sub-criterion D2 (next 50 years) is **Data Deficient.**

Historical change (D3)

The lack of graziers in this ecosystem type means that declines in vegetation due to livestock grazing or planned burns are unlikely (Zylstra 2006). However, data were lacking on change in vegetation cover in *Fjaeldmark* since European colonisation as impacts occurred before the available records. The risk status is **Data Deficient** under sub-criterion D3

Criterion E: Quantitative analysis of probability of collapse

Summary

The major future threats to *Fjaeldmark* are shrub encroachment, loss of periglacial processes and fire. However, there are insufficient data are available to quantitatively estimate whether the *Fjaeldmark* will collapse within the next 50 to 100 years. The risk status is **Data Deficient** under criterion E.

Fjaeldmark dissected by a walking track near Mt Northcote, Kosciuszko National Park. Photo: Keith McDougall.

Snowpatch herbfield

Authors

Jessica A. Rowland, Brodie Verrall, Emma Sumner, Susanna Venn, Chloe Sato, Jamie Kirkpatrick, Tracey Regan

Reviewers

Arn Tolsma, Emily Nicholson

Contributors

Matt White, Michael Traurig, Catherine Pickering, Ken Green, Casey Gibson, Ty Karpala, Annette Muir, Jayne Balmer, Genevieve Wright, Arn Tolsma, Tracey Regan, Kate Umbers

Biome

T6 Polar-alpine

Functional group

T6.4 Temperate alpine grasslands and shrublands

IUCN status

Endangered (Vulnerable-Endangered)

Assessment Summary

Australian *Snowpatch Herbfield* is restricted to high mountains and occurs in landscape settings where winter snowfall accumulates, and the snowpack persists into the summer months. Our assessment showed that the ecosystem type is **Endangered (Vulnerable-Endangered)** due to its restricted geographical distribution, and ongoing declines associated with reduced snow persistence and shrub encroachment (Table 24). Our analyses revealed considerable uncertainty in estimates of risk based on some criteria, particularly those related to thresholds of collapse caused by biotic change.

Table 24. Summary of the Red List of Ecosystems assessment of the Australian *Snowpatch Herbfield.* Category ranges in parentheses reflect uncertainty in assessment under the corresponding criteria.

Criteria	**A**	**B**	**C**	**D**	**E**	**Overall**
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	LC	VU	VU (NT-VU)	DD	**EN** **(VU-EN)**
Sub-criterion 2 *A, C, D: 50-year period including present & future* *B: AOO*	NT (LC-EN)	VU	EN (VU-EN)	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	DD	VU	DD	LC (LC-VU)		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria.

Ecosystem Description

Ecosystem Classification

In the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022a), Australian *Snowpatch Herbfield* is a sub-global ecosystem type (Level 6) belonging to Ecosystem Functional Group T6.4 Temperate alpine grasslands and shrublands, within the Polar/alpine (cryogenic) biome. The vegetation conforms to the 'herbfield' class of Specht (1981).

Australian *Snowpatch Herbfield* includes a number of structural and floristic types including 'tall alpine herbfield', 'short alpine herbfield', 'short turf', 'feldmark', 'alpine heath', 'bolster heath', 'mat heath', 'alpine sedgeland' (Gibson & Kirkpatrick 1985; Kirkpatrick 1997; Costin et al. 2000; Wahren et al. 2001; McDougall & Walsh 2007; Venn & Morgan 2007; Green & Pickering 2009b, 2009a; Parry et al. 2016; Venn et al. 2017). However, the defining characteristic is the frequent persistence of snow into summer and consequently a relatively short growing season.

There is formal recognition of this ecosystem type under relevant threatened species and communities legislation in two of three Australian states in which it occurs. In Victoria, it is listed under the Flora and Fauna Guarantee Act 1988 (Government Gazette G 27, published on 5 July 2012) and in New South Wales, 'Snowpatch herbfields in the Australian Alps Bioregion' is listed as a critically endangered ecological community under the Biodiversity Conservation Act 2016 (Listed on 27 April 2018). *Snowpatch Herbfield* is not currently listed as a threatened native vegetation community in Tasmania under the Nature Conservation Act 2002. Nor is it currently recognised under the Commonwealth Environmental Protection and Biodiversity Conservation Act 1999. However, listing is undertaken for ecological communities nationally and in all Australian states following an ad-hoc public submission process, rather than following a systematic approach.

Distinction from similar ecosystem types

Snowpatch Herbfield can have a similar plant species composition and structure as adjacent ecosystem types such as *Alpine-subalpine Open Grassy Heathland and Herbfield* and *Alpine-subalpine Closed Heath*. *Alpine-subalpine Closed Heath* is defined by a dense canopy of scleromorphic shrubs that are fire-tolerant and with typically rocky substrates (Williams & Ashton 1988). *Alpine-subalpine Open Grassy Heathland and Herbfield* occurs on shallow slopes at higher elevations and is characterised by forbs, grasses, and only a short, often prostrate dwarf shrub canopy (0.1-0.5 m; 0-30% cover) of predominantly non-resprouting (obligate seeding) shrub species (Williams & Ashton 1988). *Snowpatch Herbfield* is unique in that it occurs in areas where winter snowfall accumulates and the snowpack persists into summer, resulting in a shorter growing season than in adjacent areas (Wahren et al. 2001; Green & Pickering 2009b, 2009a; Venn et al. 2011).

Distribution

Snowpatch Herbfield occurs in the high mountain environments in south-eastern Australia (New South Wales, Victoria and Tasmania; Figure 47). It spans a latitudinal range of c. 6°, occurring between 145.93° and 148.42° longitude and between -36.15° and -43.50° latitude. The altitudinal band occupied by the ecosystem type is between approximately 1500 m and 2200 m above sea level on the mainland, and between 1010 m and 1440 m in Tasmania (Kirkpatrick et al. 2017). *Snowpatch Herbfield* occurs in both the alpine and the treeless, high subalpine zones of the Australian Alps (Williams et al. 2006, 2014; McDougall & Walsh 2007). This includes Kosciuszko National Park in New South Wales (Costin et al. 2000; Green & Pickering 2009b, 2009a) and the Bogong High Plains and surrounding peaks in Victoria (Wahren et al. 2001). In Tasmania, the ecosystem type occurs in the Central and Western highlands of Tasmania (Gibson & Kirkpatrick 1985). *Snowpatch Herbfield* exists where snow lasts well into the summer months, the result of the interplay among topography, the predominant direction of snow-bearing winds, and aspect. *Snowpatch Herbfield* has a combined known extent of 6.27 km^2.

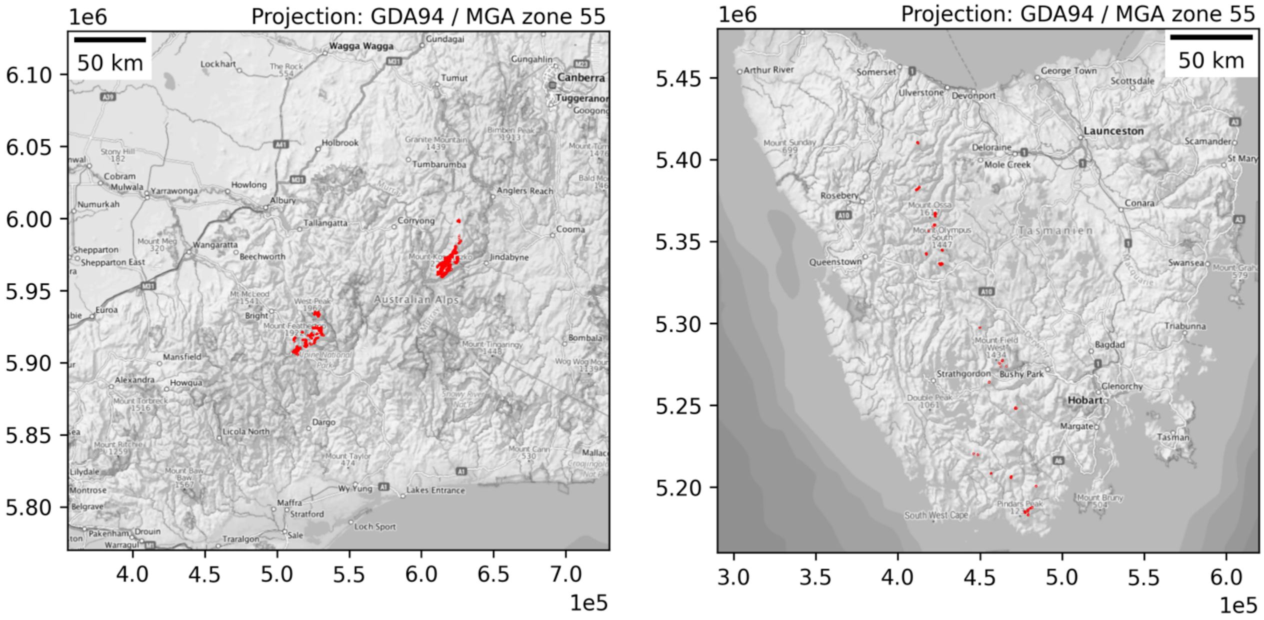


Figure 47. Distribution of *Snowpatch Herbfield* (red) across the Australian mainland (left) and Tasmania (right).

Abiotic environment

Australian alpine and subalpine landscapes are characterised by relatively low annual temperatures and high annual precipitation compared to the surrounding landscape (Williams et al. 2006, 2014). The fundamental environmental determinant of the distribution of *Snowpatch Herbfield* is the persistence of late-lying snow (Figure 48) (Wahren et al. 2001; Green & Pickering 2009b). *Snowpatch Herbfield* in south-eastern Australia occurs on lee slopes (south to south-east aspects on the mainland, and on north-east to east slopes in Tasmania) of the predominant snow-bearing winds. *Snowpatch Herbfield* can often occur in conjunction with cold-climate landforms that are a result of the glacial and peri-glacial conditions of the recent past, such as terraces, solifluction lobes, glacial cirques and stony pavements (Williams et al. 2006). Snow-shearing processes on very steep slopes can result in soil erosion of the shallow, rocky soils particularly in the Snowy Mountains (Green & Pickering 2009b). On the mainland, incident radiation during the spring and summer is reduced given their southerly aspect (Williams & Ashton 1987; Costin et al. 2000; Wahren et al. 2001; Edmonds et al. 2006; Green & Pickering 2009b).

In Tasmania, *Snowpatch Herbfield* is often the only ecosystem type with snow mid-summer (Parry et al. 2016). Snow can persist well into the alpine and subalpine growing season on the mainland (November to February, depending on altitude), and into January in Tasmania (J. Kirkpatrick, pers. Comm., 2020), resulting in a shorter growing season than in adjacent areas (Wahren et al. 2001; Green & Pickering 2009b, 2009a; Venn et al. 2011). For example, in the Kosciuszko region, between 2003–2004 and 2005–2006, the snow-free period of the centre of large *Snowpatch Herbfield* was between 53 and 76 days, compared with a snow-free period of 147–159 days in adjacent tall alpine herbfield (Green & Pickering 2009b).

Figure 48. Patches of late-lying snow in *Snowpatch Herbfield* on the Main Range, Kosciuszko National Park, NSW, at elevation of ca. 2000 m, in December 2009; View from Carruthers Peak. Photo: Susanna Venn.

Characteristic native biota

Vegetation cover in *Snowpatch Herbfield* can vary substantially, both within and between individual patches, from near complete to < 10%; bare ground and rock cover is similarly variable (Green & Pickering 2009a). *Snowpatch Herbfield* is also floristically variable both between and within patches and differ significantly between the mainland and Tasmania (Parry & Balmer 2017). Floristic variation within *Snowpatch Herbfield* (upper, mid and lower zones; central and outer zones) has been documented and typically corresponds to differences in snow duration, soil depth, and mean soil temperatures (Atkin & Collier 1992; Wahren et al. 2001; Green & Pickering 2009b, 2009a; Venn et al. 2011).

Snowpatch Herbfield is typically dominated by short graminoids (*Carex* spp., Cyperaceae; *Luzula* spp., Juncaceae; *Rytidosperma* spp., *Agrostis* spp.; Poaceae), short forbs such as *Montia* (Portulacaceae), *Brachyscome* spp., *Cotula*, *Ewartia* (Asteracea), in some situations by taller forbs such as *Celmisia* spp. (Asteraceae) and tussock grasses (e.g., *Poa* spp) (Figure 49). *Plantago glacialis* occurs in seepage areas beneath *Snowpatch Herbfield* in the Bogong High Plains (where it is extremely rare) and also in Tasmania and the Kosciuszko region. Shrubs are typically rare or absent, especially at high altitudes in the Kosciuszko region (Green & Pickering 2009a, 2009b).; species include dwarf shrubs such as *Melicytus* sp. (Violaceae) and *Coprosma niphophila* (Rubiaceae). By contrast, on the Bogong High Plains, tall shrubs (up to 50 cm high) that dominate *Alpine-subalpine Open Grassy Heathland and Herbfield* on adjacent slopes may also be present within *Snowpatch Herbfield*, but typically with < 10% cover in mid-late snowmelt areas. These shrub species include *Acrothamnus montanus* (Ericaceae) and *Grevillea australis* (Proteaceae) (S. Venn, pers. comm., 2019). Shrubs can occur at similar low densities in Tasmanian extents (Figure 49), including *Dracophyllum minimum*, *Gaultheria depressa* (Ericaceae), and *Ozothamnus rodwayi* (Asteraceae) (J. Kirkpatrick, pers. comm., 2019).

Data on *Snowpatch Herbfield* fauna are sparse. Collections in 2011-2012 from pitfall traps revealed 118 species/morphospecies, from six classes, 17 orders and 61 families; only 45 have been positively identified to genus/species across all fauna groups (M. Nash, unpublished data, 2014). An undescribed spider from the genus *Micropholcomma* is thought to be a *Snowpatch Herbfield* species. In Tasmania, wallabies selectively graze *Snowpatch Herbfield* vegetation heavily, maintaining low-structure vegetation (Gibson & Kirkpatrick 1985; Roberts et al. 2011). Functionally important arthropod groups are millipedes, mites, spiders, springtails, beetles and ants (Green & Osborne 2012). This represents a wide range of trophic levels, from consumers to predators. The suite of invertebrates probably plays a significant role in the maintenance of soil health (Jouquet et al. 2006; Paoletti et al. 2007).

In Tasmania, *Snowpatch Herbfield* contains a suite of obligate seeding species including *Colobanthus pulvinatus*, *Plantago glacialis* and *Gaultheria depressa* (Parry & Balmer 2017). On the mainland, most plants and invertebrates recorded in *Snowpatch Herbfield* also occur in other alpine vegetation communities across *Alpine-subalpine Open Grassy Heathland and Herbfield* (Wahren et al. 2001; Venn & Morgan 2007; M. Nash, unpublished data, 2014). However, in the Kosciuszko region, *Coprosma niphophila* and *Colobanthus nivicola* are restricted to *Snowpatch Herbfield* (Edmonds et al. 2006; Green & Pickering 2009b, 2009a). Despite the apparent absence of specialist *Snowpatch Herbfield* species on the mainland, the community is distinctive (Williams et al. 2006; McDougall & Walsh 2007). There are compositional differences between *Snowpatch Herbfield* as a function of altitude, both with respect to plants (McDougall 1982; Wahren et al. 2001) and invertebrates (M. Nash, unpublished data, 2014). There are also gradients in floristic composition between snowmelt zone, upper slope, mid-slope and lower slope regions of *Snowpatch Herbfield* (Atkin & Collier 1992; Wahren et al. 2001; Venn & Morgan 2007; Green & Pickering 2009a; Venn et al. 2011; Parry et al. 2016).

Figure 49. *Snowpatch Herbfield* on Club Lake, Kosciuszko National Park, New South Wales. Photo: Susanna Venn.

Key components, processes, and interactions

Snow persistence

The principal determinant of the distribution of *Snowpatch Herbfield* is the variation in the persistence of snow on high elevation, lee-side ridges (Costin et al. 2000; Williams et al. 2006, 2014; Kivinen et al. 2012) (Figure 50). Subsequently, variation in snow-lie persistence is determined by the amount of snowpack in the previous winter and how quickly the snow melts in the following spring and summer (Green & Pickering 2009b). Short growing seasons resulting from delayed snowmelt and very cold temperatures largely restricts the prevalence of shrub species but supports vegetation dominated by short herbs and graminoids (Venn et al. 2011; Williams et al. 2015). Snow insulates the vegetation against exposure to extremely cold temperatures and frosts (Billings & Bliss 1959). Snow persistence in *Snowpatch Herbfield* may be highly variable between years. Over the past 50 years in the Kosciuszko region, 21 ha of land is estimated to have supported persistent summer snowdrifts in 80% of years, while another 440 ha supported persistent summer snowdrifts in 10% of years (Edmonds et al. 2006).

Soils and geomorphic processes

Mainland snowdrifts that persist into late spring and summer accumulate windblown sediments that add to the soil matrix of *Snowpatch Herbfield* (Costin et al. 2000) and release nitrogen into the underlying vegetation during the growing season (Bowman 1992). Aeolian sediments also decrease surface albedo, leading to an increased rate of snowpatch ablation (K. Green, pers. observ.). The moisture regime of *Snowpatch Herbfield* varies with position with downslope sites receiving more spring run-off than upslope sites. This moisture gradient has been correlated with vegetation composition (Atkin & Collier 1992; Green & Pickering 2009a). Deep snow on steep slopes can also generate substantial sheer force (Costin et al. 1973) which can cause local disturbance in the form of vegetation removal, soil movement and the creation of bare ground. *Snowpatch Herbfield* provides important late-season food resources for a range of invertebrate fauna. In Tasmania, macropods maintain the characteristic low-statured vegetation by selective grazing during the snow-free season (Gibson & Kirkpatrick 1985).

Figure 50. Conceptual model of the key features, interactions, processes and threats in *Snowpatch Herbfield*.

Major threats

Climate change

Snowpatch Herbfield is likely to be highly sensitive to climate change, through the indirect effects of temperature and precipitation affecting snowfalls and snowpack, and through invasion/encroachment of shrubs and tall grasses as the growing season extents. Persistent snow is a key driver of *Snowpatch Herbfield* communities, as short growing seasons maintain dominance of low-statured graminoids and forbs. Declines in the amount and longevity of snow cover will reduce the strength of this important environmental filter (Venn et al. 2011). Increasing temperatures and declining snowfall have led to a 30% reduction in snow cover over the past 60 years on the Australian mainland (Davis 2013; Pepler et al. 2015). In the future, snow duration is predicted to reduce by approximately 100 days and maximum depths to decrease to < 10% of their present value under a high impact scenario for 2050 (Hennessy et al. 2003). In contrast to the mainland, annual snow incidence fluctuated between years with no clear trend over three decades at Mt Field, as snow incidence is affected strongly by wind speed (Kirkpatrick et al. 2017).

Reductions in snowfall and snowpack longevity, and increasing temperatures during the growing season, will favour the expansion of taller shrubs (Venn et al. 2011; Wahren et al. 2013; Williams et al. 2014; Camac et al. 2015) changes to the vegetation composition (Pickering et al. 2014). Declining snow persistence and associated soil-drying will also likely reduce the habitat suitability for many characteristic plant species, as late-lying snow provides a source of meltwater, keeps soils moist, and therefore provides suitable conditions for species restricted to the lower or wetter zones of *Snowpatch Herbfield* including *Plantago glacialis* and *Montia australasica* (Green & Pickering 2009a). The effects of soil drying will also impact seedling establishment and reduce late-season food sources for invertebrates and marsupials in Tasmania (Kirkpatrick, pers. comm., 2019). Loss of snow will expose the vegetation to more variable climatic conditions, which may lead to greater frost exposure (Venn et al. 2009).

Fire

Fire occurs infrequently in *Snowpatch Herbfield* but can strongly impact vegetation. For example, several patches in Tasmania show legacy effects of fires in the 1960s and 1980s, including increased cover of shrubs and cushion plants (Parry et al. 2016; Kirkpatrick et al. 2017). *Snowpatch Herbfield* vegetation is not typically flammable, whereas shrubs are highly flammable (Fraser et al. 2016) due to the oil in their leaves and structure. Climate change and altered fire regimes may further enhance the encroachment of shrubs, and the encroachment of shrubs may alter fire regimes (Camac et al. 2015; Williams et al. 2015).

Alien ungulates and plants

Horse and deer numbers have increased substantially in the mainland over the past decade (Nimmo & Miller 2007; Dawson & Miller 2008; Williams et al. 2014), particularly in the Kosciuszko area where a 23% increase per annum was recorded between 2014 and 2019 (Cairns 2019). Changes to *Snowpatch Herbfield* through shrub expansion may be exacerbated on the mainland by continued grazing of exotic ungulates such as sambar deer and horses, and fallow deer in Tasmania, through selective browsing of the herbaceous vegetation and the creation of bare ground (Williams et al. 2014). Deer impact surveys have indicated that deer use *Snowpatch Herbfield* in the snow-free season more than other low-statured plant communities, such as *Alpine-subalpine Open Grassy Heathland and Herbfield*, and have been recorded grazing on forbs and grasses including *Celmisia* spp., *Poa* spp., and *Psychrophila intraloba*. Destructive trampling by sambar deer of the snowpatch-associated *Alpine-subalpine Open Grassy Heathland and Herbfield,* specialist, *Plantago glacialis,* has also been recorded in the Bogong High Plains (Z Walker, unpublished data, 2019).

In addition to native shrubs (Bridle et al. 2017) introduced plants, especially *Hypochaeris radicata*, appear to be increasing in abundance in *Snowpatch Herbfield* on the mainland (C.-H. Wahren et al., unpublished data, 2014; K Green, pers. comm., 2019).

Human use

Lasting snow in *Snowpatch Herbfield* is an inviting landscape feature for hikers in summer when the surrounding landscape is snow-free. Nonetheless, hikers are unlikely to significantly increase the rate of snowmelt, unless they actively remove snow by shovelling. However, in Victoria, at least two distinct areas of *Snowpatch Herbfield* have been destroyed by development of ski resorts in the last 50 years (W. Papst, unpublished data, 2014).

IUCN Stresses Classification

1.2 – Ecosystem degradation (e.g., altered temperature/precipitation regime)
2.3.2 – Indirect species effects (Competition)

IUCN Threats Classification

1.3 – Tourism and recreation areas (ski development)
6.1 – Recreational Activities (hikers, any others)
8.1.2 – Invasive non-native species (feral horses, sambar deer, fallow deer)
11.3 – Temperature extremes

Ecosystem collapse

Snowpatch Herbfield can collapse and transition into *Alpine-subalpine Open Grassy Heathland and Herbfield* due to changes in the vegetation structure and composition. This occurs when the native, short herbaceous vegetation becomes dominated by taller growing native species, such as native shrubs or taller graminoid and forb species (Venn et al. 2011; Pickering et al. 2014). The characteristic species in *Snowpatch Herbfield* are maintained by the late-lying snow limiting the growing season. Changes in species composition and structure are likely under climate change as the snowmelt date becomes more aligned with that in the surrounding landscape.

For *Snowpatch Herbfield*, we regarded ecosystem collapse as having occurred when:

- *Area*: The mapped distribution declines to zero (100% loss) (Criteria A and B), or;
- *Snowmelt date*: The date the snow melts is the same as the surrounding landscape (i.e., late-lying snow that supports the ecosystem type no longer persists (Criterion C), and/or
- Vegetation cover: Cover of native shrubs is > 25% (in New South Wales and Tasmania) or graminoid cover is > 75% (Criterion D).

Risk Assessment

Criterion A: Reduction in distribution

Summary

The risk status of *Snowpatch Herbfield* is **Least Concern** under sub-criterion A1, **Near Threatened (Least Concern-Endangered)** under sub-criterion A2, and **Data Deficient** under sub-criterion A3.

Methods

Time-series maps of *Snowpatch Herbfield* are unavailable. However, some losses of *Snowpatch Herbfield* extent are known to have occurred due to developments (Williams et al. 2015), reduced persistence of late-lying snow (Green and Pickering 2009b; Green et al unpublished data 2020; Pickering et al. 2014) and encroachment of shrubs from adjacent ecosystems (Kirkpatrick et al. 2017).

Therefore, we based the assessment of criterion A on anecdotal evidence, data on several individual patches and inferences from data on changes in snow cover and shrub cover. We used measures from Kirkpatrick et al. (2017) of the spatial extent of five sites of *Snowpatch Herbfield* in Tasmania (Black Bluff, Cradle Mountain, Barn Bluff, Mount Rufus, Hill One) from aerial photographs and/or satellite images for 2015, and an appropriate previous year for each site. We excluded the data for Hill One area estimate for the initial date (1946) was well outside our timeframe of interest for sub-criterion A1.

We inferred the likely future changes in ecosystem area (sub-criterion A2b) based on projected changes in the persistence of snow cover and shrub cover. We used projections on changes in the persistence of late-lying snow based on time-series data on snow duration in 19 sites spanning 1996 to 2019 in Kosciuszko National Park, New South Wales (Green & Pickering 2009a; Pickering et al. 2014; Green et al unpublished data 2020) (see criterion C for full details). We also used the predicted future rate of increase in shrub cover in the Bogong High Plains, Victoria, estimated by Williams et al. (2015), (see criterion D for full details).

Assessment outcome

Recent change (A1)

Over the past 50 years, no snowpatches in Tasmania (J. Balmer, J. Kirkpatrick, pers. comm., 2019) or New South Wales (K. Green, pers. comm., 2019) have been lost or significantly reduced in size from developments such as ski resorts, walking tracks or other human developments. However, data on four snowpatches in Tasmania (Black Bluff, Cradle Mountain, Barn Bluff and Mount Rufus: Kirkpatrick et al. 2017) indicate that change in snowpatch area is not consistent across the state. These data show that area has decreased by approximately 0.12% per year on average, but ranges from -0.4 (Barn Bluff) to 0.1 per year (Mount Rufus). Based on these estimates, area has changed by -6.2% on average (range: -22.4–5.4%) over the past 50 years, and thus does not meet the threshold for Vulnerable even under the worst-case scenario.

In New South Wales, the decline in snow duration (see criterion C; Green & Pickering 2009a) is likely to have caused a decline in individual snowpatch area as longer growing seasons allow for encroachment from adjacent ecosystems. However, sufficient data are not available to reliably estimate change in area at this scale. Over the past 50 years, the distribution of *Snowpatch Herbfield* in Victoria has decreased due to ski resort development. Of the 240 areas of *Snowpatch Herbfield* on the Bogong High Plains mapped by McDougall (1982), at least two have been destroyed from ski resort development in the past 50 years (W. Papst, unpublished data, 2014). This represents a decline of less than 5%. The risk status is **Least Concern** under sub-criterion A1.

Future change (A2b)

Following Williams et al. (2015) we assumed that because all *Snowpatch Herbfield* is within Australia's National Reserve System, none will be destroyed by human land use in the future. Future declines in *Snowpatch Herbfield* on mainland Australia are likely due to the indirect effects of climate change (Green & Pickering 2009). The amount and longevity of snow cover is predicted to decline over the next 50 years due to increasing temperatures and declining snowfall; Snow duration is predicted to reduce by approximately 100 days and maximum snow depths to decrease to less than 10% of their present value under a high impact scenario for 2050 (Hennessy et al. 2003).

The warmer temperatures and decreased snow duration will increase the growing season (Green & Pickering 2009a), facilitating the encroachment of adjacent shrubs and grasses (Green & Pickering 2009b; Venn et al. 2011; Wahren et al. 2013; Williams et al. 2014; Camac et al. 2015). In Victoria and New South Wales, snow cover is predicted to thaw 69.8% (54.6-88.8%; see criterion C2b) earlier by 2045, which is likely to lead to a loss of *Snowpatch Herbfield* area. In Victoria, shrub cover is predicted to increase to 62.9% by 2046 (plausible range 4–100%; see sub-criterion D2b). The ecosystem type shows strong signs of declines based on the predicted degradation from decline in snow duration and shrub encroachment, however, the likely severity of declines in area are unclear. Therefore, we cautiously estimate that the risk status is **Near Threatened (Least Concern to Endangered)** under sub-criterion A2.

Historical change (A3)

Past livestock grazing between c. 1850 and 1950 is likely to have damaged *Snowpatch Herbfield*. Livestock grazing is known to have caused substantial damage in Australian alpine environments, including *Snowpatch Herbfield* (Costin 1954; Williams et al. 2014). However, data are lacking on the total area or number of patches lost since grazing commenced. The risk status is **Data Deficient** under sub-criterion A3.

Criterion B: Restricted distribution

Summary

The risk status of *Snowpatch Herbfield* is **Least Concern** under sub-criterion B1 and is **Vulnerable** under sub-criteria B2 and B3.

Methods

Under criterion B, the extent of occurrence (EOO; B1) and area of occupancy (AOO; B2) were determined using a combination of existing map products from New South Wales (Costin et al. 1979; Ecology Australia 2003), Victoria (DELWP 2018a ; comprising mapping based on McDougall 1982), and Tasmania (Kirkpatrick & Marsden-Smedley 2014). These data were augmented by additional mapping undertaken for this analysis across regions of NSW not accounted for by existing mapping. *Snowpatch Herbfield* polygons were rendered using aerial photography flown in November 2015.

The number of threat-defined locations (sub-criterion B3) was based on encroachment and extreme temperatures.

Assessment outcome

Sub-criterion B1

The Extent of Occurrence (EOO) based on available mapping is currently estimated at 75,346.51 km^2 (Figure 51). The risk status is **Least Concern** under sub-criterion B1. Under a previous analysis of the risk to mainland *Snowpatch Herbfield* undertaken by Williams et al. (2015), the EOO was 2,971 km^2. The discrepancy between this analysis and that undertaken previously is largely due to the broad vegetation mapping unit assigned to the ecosystem type in NSW and the coarser mapping resolution of the NSW data used at the time (see Thomas et al. 2000).

Sub-criterion B2

Snowpatch Herbfield covers 33 10×10 km grid cells (Area of Occupancy, AOO) (Figure 51). There is evidence that the ecosystem type will continue to decline due to documented reductions in snow persistence and shrub incursion (see criteria C and D below). The previous AOO covered 19 10×10 km grid cells of which 6 cells contained >1% coverage of the ecosystem type (see B1 for explanation; Williams et al. 2015). The risk status of the ecosystem type in the current assessment is **Vulnerable** under sub-criterion B2.

Sub-criterion B3

The major threats to *Snowpatch Herbfield* are snow persistence and encroachment of shrubs and tall grasses. Encroachment alone is unlikely to cause collapse across the whole ecosystem distribution in < 20 years. Based on possible changes in climatic conditions, the ecosystem type may occur at between 1 and 3 locations. A future single location is possible if extreme temperature years are experienced across the entire distribution. Alternatively, there could be up the three locations, as patches are clustered in New South Wales, Victoria and Tasmania. Threats could act differently in each of these regions, due to varying climate change impacts relating to elevation. The risk status is **Vulnerable** under sub-criterion B3.

Figure 51. Map of Australian *Snowpatch Herbfield* (magenta polygons), showing EOO (black polygon) and AOO (green squares).

Criterion C: Environmental degradation

Summary

The risk status of *Snowpatch Herbfield* was assessed as **Vulnerable** under sub-criterion C1, **Endangered (Vulnerable-Endangered)** under sub-criterion C2b, and **Data Deficient** under sub-criterion C3.

Identification of abiotic indicator

We examined the relevance and data availability/quality for two indicators to assess the risk of collapse from environmental degradation:

- *Snowmelt date*: a direct measure of the longevity of snowpack in *Snowpatch Herbfield*, measured as the proportion of months of the year that snow persisted in an individual snowpatch starting in July (month 1) until all snow had melted.
- *Snowpack depth*: measures the average depth of snowpack (cm) and is a proxy for the longevity of snowpack.

Indicator: Snowmelt date

Relevance to ecosystem function

Snowpatch Herbfield is defined by the presence of late-lying snow into the growing season (Wahren et al. 2001; Green & Pickering 2009a). Short growing seasons resulting from a late snowmelt date encourage the dominance of short forbs and graminoids that define the ecosystem type and impede the encroachment of shrubs from adjacent ecosystems (Venn et al. 2011; Williams et al. 2015). Late-lying snow is an important source of meltwater and maintains high soil moisture levels in some areas. Significantly earlier snowmelt will likely cause these wetter areas across the ecosystem type to dry out much earlier, potentially reducing the suitability of *Snowpatch Herbfield* for many characteristic plant species.

Data availability and quality

Recent change (C1)

For New South Wales, we combined two time-series datasets to calculate the number of months from 1st July each year until the snow melted from 1996 until 2019 (Figure 52). Green et al. (unpublished data, 2020) measured the 19 longest lasting areas of *Snowpatch Herbfield* in Kosciuszko National Park, New South Wales, between 1996 and 2019. Where all snow had melted before 1st January, the month of snowmelt was recorded as December. Green & Pickering (2009b) and Pickering et al. (2014) recorded the thaw dates of seven of the longest persisting areas of *Snowpatch Herbfield* in the Snowy Mountains, New South Wales after the winters of 1996, 1999, 2001, and 2003 to 2012. These data included the thaw date of five zones within each snowpatch, of which we selected the latest thaw date among zones in each site. Where there were conflicting snowmelt dates across these datasets, we conservatively used the later date.

Data on snow duration in Victoria are unavailable. Yet the observed increase in average winter temperature is a known driver of declines in snow depth and duration (Davis 2013; Sánchez-Bayo & Green 2013). The annual average maximum winter temperatures in Victoria and New South Wales were highly positively correlated (0.983). Therefore, we assumed the trend in snow duration observed in New South Wales is approximately representative of declines in Victoria. For Tasmania, we used the analysis of Landsat images from Kirkpatrick et al. (2017) of Mt. Field National Park. Data show the change in percentage of clear days with snow present between April and December between 1983 and 2014 to indicate change in snow persistence. To estimate snowmelt date in adjacent alpine ecosystems, we used snow depth data from 1969 to 2019 at Spencers Creek in Kosciuszko National Park, New South Wales (available from https://www.snowyhydro.com.au/our-energy/water/inflows/snow-depths-calculator/) (Figure 52). We assumed that the snow in Spencers Creek had melted when the depth was 0 cm.

Future change (C2)

We extrapolated the trend in snow duration in New South Wales (see sub-criterion C1) to estimate the trajectory of decline for *Snowpatch Herbfield* and for the Spencers Creek site.

Historical change (C2)

Data were unavailable to assess historical change in snowmelt date.

Selection of collapse threshold

Snowpatch Herbfield is characterised by snow persisting for longer than it persists in the adjacent ecosystems. Therefore, we assumed the ecosystem type is collapsed when the average month of snowmelt across all monitored snowpatches is the same as the month of snowmelt in Spencers Creek, New South Wales, as a proxy for the surrounding landscape (i.e., the difference between Spencers Creek and *Snowpatch Herbfield* is 0).

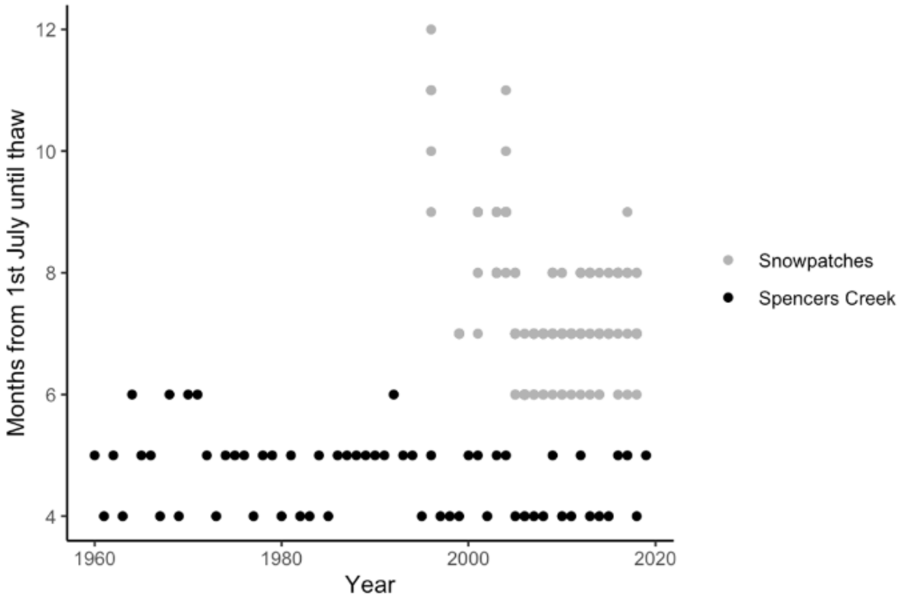


Figure 52. Snowmelt duration in Kosciuszko National Park, New South Wales in in *Snowpatch Herbfield* (Green & Pickering 2009b; Pickering et al. 2014; Verrall 2023) and at Spencers Creek (https://www.snowyhydro.com.au/our-energy/water/inflows/snow-depths-calculator/). Data are the number of months from 1st July until the snowpatch fully thawed.

Calculation of initial and present/future values

Recent change (C1) and Future change (C2)

For snowmelt date, data were converted to the proportion of the year that snow persisted based on the number of months between July and the month of melting. We fitted a quasi-binomial hierarchical generalized linear model to the data, with year as a fixed effect and survey site as a random effect. We used the model to extrapolate from 1996 back to 1969, and from 2019 to 2045. Predicted means for 1969 to 2019 are presented in Figure 53 with error bars representing the modelled standard errors. To measure declines over the past 50 years (C1), we used the modelled value in 2019 as the present value, and1970 as the initial value. To estimate future declines over any 50-year period (C2b), we used the modelled value in 1995 as the initial value, and 2045 as the future value.

There are inherent uncertainties in predicting future environmental conditions, and snow is certainly one of the most challenging environmental features to robustly project into the future. However, many climate projection studies (Harris et al. 2016; Di Luca et al. 2018) indicate that the near-surface temperatures are going to rise which will negatively affect snowpack characteristics (including snow depth, snow cover, snow duration). In particular, Di Luca et al. (2018) suggests there will be considerable declines in snow depth (and other features) of 15% and up to 60% by 2030 and 2079, respectively. Comparably, our extrapolation estimates an average decline of ~17% (range: 10-24%) between 2020 and 2039. Therefore, it is reasonable to assume a decline in snow depth, cover and duration to occur. We apply plausible bounds for the risk category to account for the uncertainty in the extrapolation.

We also converted the Spencers Creek data to the proportion of the year that snow persisted. We fitted a quasi-binomial generalized linear model to the data, with year as a fixed effect. We used the model to extrapolate from 2019 to 2045. Predicted means for 1951 to 2019 are presented in Figure 53. with error bars representing the modelled standard errors. To measure recent change (C1), we used the value in 2019 as the present value, and 1969 as the initial value. To measure future change (C2b), we used the modelled value in 1995 as the initial value and 2045 as the future value

Historical change (C3)

Data were unavailable to assess this sub-criterion.

Calculation of severity and extent

Recent change (C1)

In Tasmania, there was some decline in snow incidence on low elevation mountains, and slight increases on high elevation mountains (Kirkpatrick et al. 2017). Kirkpatrick et al. (2017) suggest the weather systems that bring snow to Tasmania have strengthened in recent decades, and that clear skies and an associated reduction in spring rainfall may be enabling snow persistence. Overall, there was no evidence of a decline in the length of snow lie between 1983 and 2014 (Kirkpatrick et al. 2017). In New South Wales, the proportion of months with snow cover in 1969 was estimated as 0.4063 (range from SE bars: 0.3961-0.4165) in Spencers Creek and 0.8131 (range of SE bars: 0.7742-0.8520) in *Snowpatch Herbfield*, resulting in a difference of 0.4068 (Figure 53). In 2019, the proportion of months with snow cover was estimated as 0.3641 (range: 0.3514-0.3768) in Spencers Creek and 0.5622 (range: 0.5505-0.5739) in *Snowpatch Herbfield*, a difference of 0.1981. The relative severity of decline in snow persistence is 100 × (0.4068 - 0.1981)/(0.4068 – 0) = 51.3%. The lower and upper bounds of decline in snow persistence are 100 × (0.3577 - 0.1737)/(0.3577 – 0) = 51.4% and 100 × (0.4559 - 0.2225)/(0.4559 – 0) = 51.2%. Assuming that these data are representative of patterns in New South Wales and Victoria, which cover 50-80% of the ecosystem distribution, the risk status is **Vulnerable** under sub-criterion C1.

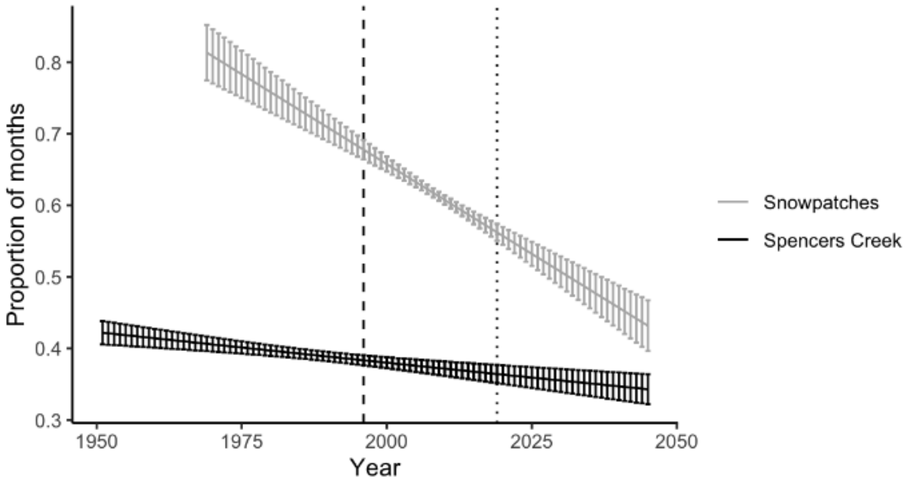


Figure 53. Modelled data on snowmelt duration in *Snowpatch Herbfield* and Spencers Creek, Kosciuszko National Park, New South Wales. Data are the proportion of months from 1st July until the snowpatches fully thawed. Dashed line is the first year for which there are data for snowpatches and dotted line is the last year there are data for *Snowpatch Herbfield* and Spencers Creek; data are extrapolated beyond these years. Error bars are the modelled predictions' standard error.

Future change (C2b)

Snow cover and duration have been projected for several potential future climates for the Climate Futures for the Australian Alps project (Harris et al. 2016). Despite variation among the six climate models, these projections indicate that snow volume and duration may decline and the area of the alps covered with snow may contract to only the highest peaks by the end of the 21st century. However, these data were unavailable for detailed analysis in the present assessment. We extrapolated the trend in snow duration in New South Wales to estimate the trajectory of decline between 1995 and 2045 for *Snowpatch Herbfield* and for the Spencers Creek site (Figure 53). In 1996, the proportion of months with snow cover was 0.3833 (range: 0.3763-0.3903) in Spencers Creek and 0.6776 (range: 0.6679-0.6973) in *Snowpatch Herbfield*, a difference of 0.2943. In 2045, snow duration estimated as 0.3429 (range: 0.3220-0.3638) in Spencers Creek and 0.4317 (range: 0.3962-0.4672) in *Snowpatch Herbfield*, a difference of 0.0888. The relative severity of the predicted decline in snow persistence over the whole ecosystem type is 100 × (0.2943 - 0.0888)/(0.2943– 0) = 69.8% (plausible bounds: 54.6-88.8%). We assume these data are representative of patterns in New South Wales and Victoria, which cover 50-80% of the ecosystem distribution. Therefore, the risk status is **Endangered (Vulnerable-Endangered)** under criterion C2b.

Historical change (C3)

There were insufficient data to assess historical changes in snowmelt date. The risk status is **Data Deficient** under sub-criterion C3.

Indicator: Snowpack depth

Relevance to ecosystem function

'Snowpack' describes a quantity of fallen snow that has become massed together. Snowpack accumulates over the winter to a season high, before it begins to melt and decline. Depth of snow is one commonly measured variable of snowpack, as is snow density and duration. Deep and densely packed snow is likely to last longer in *Snowpatch Herbfield* than the surrounding landscape due to the shelter that lee-side and south-eastern ridges (on the mainland) provide. The peak depth of snowpack is only an indirect indication of when all the snow in the area might melt, which depends on many other environmental and climatic variables. Green & Pickering (2009b) indicated that there was a significant positive correlation between date of thaw of late-lying *Snowpatch Herbfield* in the Snowy Mountains and snow depth at a long-term snow monitoring site at Spencers Creek (1841 m above sea level); later thaw dates tended to be observed in *Snowpatch Herbfield* when snow depth was higher at Spencers Creek.

Data availability and quality

Due to a lack of snow depth data across the distribution of the ecosystem, we could not adequately assess this indicator. Further, snow depth is a proxy for snowmelt date, which we have analysed.

Criterion D: Disruption of biotic processes and/or interactions

Summary

The status of *Snowpatch Herbfield* is assessed as **Vulnerable (Near Threatened–Vulnerable)** under sub-criterion D1, **Data Deficient** under sub-criterion D2, and **Least Concern (Least Concern–Vulnerable)** under sub-criterion D3.

Identification of biotic indicator

We examined the relevance and data availability/quality for two biotic indicators to assess the risk of collapse from disruption of biotic processes and/or interactions:

- *Native shrub or graminoid cover:* a direct measure of the proportion of ground cover of native shrubs or snowgrass (*Poa costiniana*).
- *Phenology:* timing or length of the growing season.

Indicator: Native shrub or graminoid cover

Relevance to ecosystem function

Snowpatch Herbfield is characterised by dominance of low-statured herbs, forbs and graminoids. The proportion of low-statured plants, bare ground, rock, and tall shrub cover within and between *Snowpatch Herbfield* areas can vary greatly (Green & Pickering 2009a). However, tall shrubs from adjacent heathlands may be present in *Snowpatch Herbfield* in the Snowy Mountains at low cover of typically <10% cover (Green & Pickering 2009a). In the Victoria distribution, shrubs are common in the early snowmelt areas (up to 60% cover), but uncommon in mid and late snowmelt areas, generally less than 20% cover. An increase in native shrub cover or native grass cover would alter the characteristic structure and composition of *Snowpatch Herbfield and* ultimately cause a shift to an *Alpine-subalpine Open Grassy Heathland and Herbfield.*

Snowpatch Herbfield is also characterised by a dominance of low-statured herbs, forbs and graminoids. An increase in native grass cover, particularly snowgrass *(Poa costiniana)* would alter the characteristic structure and composition of *Snowpatch Herbfield* and ultimately cause a shift to an *Alpine-subalpine Open Grassy Heathland and Herbfield.*

Data availability and quality

Recent change (D1) and Future change (D2)

T. Karpala (unpublished data, 2020) measured the changes in the percentage of native shrub cover in 2000, 2011, 2016, and 2020 across Warby Corner (south facing) in the Bogong High Plains, Victoria. Shrub cover was estimated in 217 quadrats, 5 metres apart, across 7 transects. Measures of shrub cover in 1969 (the requisite 50 years prior to our assessment) in Victoria are unavailable, so we conservatively based our estimate on records of shrub cover in the much more recent survey data from 1995/1996 (Wahren 1997). Wahren (1997) measures shrub cover in 33 snowpatches in 1995/1996 based on 15 45 x 6 m^2 quadrats per site. A second dataset on shrub cover is available in Victoria; Wahren et al. (2001) and Williams et al. (2014) measured changes in native shrub cover in 23 *Snowpatch Herbfield* patches (500 point quadrats each) in 1996 and 2012 in the Bogong High Plains. Cover was estimated at each site from $1m^2$ point quadrats along multiple transects (Wahren et al. 2001).

Time-series data of Tasmanian vegetation cover (Figure 54) were analysed by Kirkpatrick et al. (2017). The surface cover of tall shrubs at Mt Field was estimated from quadrats in 1983, 1998 and 2014. Shrub cover at Black Bluff, Cradle Mountain, Barn Bluff and Mount Rufus were estimated from aerial photographs and/or satellite images for 2015, and a previous year (which varied among sites).

Anecdotally, encroachment by shrubs is less of a threat in New South Wales than encroachment by graminoids. Therefore, for New South Wales, we assessed change in graminoid cover. The invasion of tall grasses is of greater threat to *Snowpatch Herbfield* than encroachment of shrubs in New South Wales (Pickering et al. 2014), whereas shrub encroachment is the greater threat in Victoria and Tasmania (see below). Therefore, change in grass cover was only assessed for the New South Wales distribution. Pickering et al. (2014) and Verrall et al. (2023) measured percentage cover of graminoids in the characteristic zones (A: snowpatch feldmark; B: transition; C: snowpatch herbfields) of 7 snowpatches in Kosciuszko.

Historical change (D3)

Historical data on vegetation were unavailable. We therefore used experts' judgements to estimate the historical vegetation cover change.

Selection of collapse threshold

Through consultation with alpine experts, it was assumed the transition from *Snowpatch Herbfield* to *Alpine-subalpine Open Grassy Heathland and Herbfield* occurs when shrub cover reaches 25% or when graminoid cover reaches 75% (in NSW only).

Calculation of initial and present/future values

Recent change (D1)

We only assessed shrub cover in Tasmania and Victoria. We used the rate of change in shrub cover in Victoria estimated in Williams et al. (2015) to estimate shrub cover in 1969 and 2019. To estimate the current value of shrub cover in Victoria, we used the data collected by T. Karpala (unpublished data, 2020). While these data only cover one *Snowpatch Herbfield*, we assume it is representative of *Snowpatch Herbfield* across Victoria (S. Venn, pers. comm.). Wahren (1997) found that the mean shrub cover across 33 snowpatches was 5%, with a median of 2% and 0% (<1%) being the most frequently recorded cover of shrubs. Therefore, we conservatively estimate the initial value as 2% (range: 0-5%).

To measure changes in shrub cover in Tasmania over the past 50 years (1969-2019), we fit a linear model to the Tasmanian shrub cover data, with year as a fixed effect and site as a random effect. We examined the influence of each observation on the model using Cook's distance and determined that the values for Mount Field had a strong influence on the model. Therefore, we ran the linear model both with and without the values for Mount Field and used these models to estimate the initial value in 1969 and current value in 2019.

To measure changes in graminoid cover in New South Wales over the past 50 years (1969-2019), we fit a linear model to the New South Wales graminoid cover data, with year as a fixed effect and snowpatch site as a random effect. We used the model estimate for the current value in 2019 and the estimate of graminoid cover in 1970 from Wimbush & Costin (1979) as the initial value. Wimbush & Costin (1979) reported graminoid cover in eroded (~30%) and non-eroded (~43%) in the short herbfields zone of *Snowpatch Herbfield*.

Figure 54. Shrub cover of *Snowpatch Herbfield* in Tasmania from Kirkpatrick et al. (2017).

Future change (D2b)

To forecast future changes in shrub cover (1996–2046), we assumed that the rate of invasion observed between 1996 and 2012 in Victoria would remain constant (Williams et al. 2015).

Historic change (D3)

We followed the approach of Williams et al. (2015) to estimate the relative severity of change since 1750 in Victoria. We assumed that all shrub invasion in 2011 had occurred since 1750, and that shrub cover in 1750 was bounded between c. 1% and the lowest of bounded estimate observed in 1996 (c. 3%). We used the mean and 90% confidence intervals to calculate a best estimate, and upper and lower bounded estimates. In Tasmania, we assumed that all shrub invasion in 2019 occurred since 1750.

Calculation of severity and extent

Recent change (D1)

In Tasmania, shrub cover did not significantly change over the past 50 years based on the model with all values ($t_4 = 0.906$, $P > 0.05$) nor the model excluding outliers ($t_3 = -0.332$, $P > 0.05$; Figure 54). In Victoria, shrub cover increased by 1.6% between 2000 (6.05%) and 2020 (7.65% ± 3.79%; mean ± standard error), despite some areas being burned in a fire in 2003 (T. Karpala, unpublished data, 2020). Shrub cover was estimated as 2% (plausible range: 0-5%) in 1969. We estimated the relative severity of increase in shrub cover in Victoria to be 100 ´ (2 – 7.65)/(2 – 25) = 24.6%. We estimated the lower bound of relative severity as 100 ´ (5 – 7.65)/(5 – 25) = 13.3% and the upper bound as 100 ´ (0 – 7.65)/(0 – 25) = 30.6%. Based on the initial assessment of Victorian *Snowpatch Herbfield* (Williams et al. 2015), there was a net increase in shrub cover of 7.7 ± 3.7% between 1996 and 2012. This equates to an annual increase of 0.48% per year (0.25-0.71%) and a relative severity over the past 50 years of 77% based on the collapse threshold of 25% cover.

In New South Wales, we estimated a net increase in graminoid cover. In New South Wales, graminoid cover increased from an estimated 30-43% to 59% over the past 50 years. The estimated relative severity of change in graminoid cover ranges between 100 ´ (43 – 59)/(43 – 75) = 51% and 100 ´ (30 – 59)/(30 – 75) = 65%. The estimates for the initial value of graminoid cover are based on zone C in *Snowpatch Herbfield*, which typically has higher cover of graminoids than other areas within a snowpatch. Therefore, use of these values may underestimate the increase in graminoid cover. Based on the changes in Victoria (relative severity =

24% [13.3-30.6%] over 34% of whole distribution) and New South Wales (relative severity = 51-65% over 55% of the distribution), the risk status is **Vulnerable (Least Concern–Vulnerable)** under D1.

Future change (D2b)

Due to uncertainty in the trend for the Tasmanian extent, we were unable to reliably estimate future change. To forecast future changes (1996–2046), we assumed that the rate of invasion observed in Victoria between 1996 and 2012 would remain constant (following Williams et al. 2015). Assuming this trend will continue until at least 2046, the mean shrub cover was estimated to expand to 62.9% (plausible range 4–100%). The risk status of the ecosystem type under criterion D2 (next 50 years) was **Data Deficient,** because the outcomes spanned all risk categories (Least Concern–Critically Endangered).

Historic change (D3)

Assuming that all shrub invasions had occurred since European settlement across all Australian *Snowpatch Herbfield*, the risk status is **Least Concern (Least Concern–Vulnerable)** under sub-criterion D3.

Indicator: Phenology

Relevance to ecosystem function

The phenology in *Snowpatch Herbfield* is driven by the presence of late-lying snow and the rate of snowmelt. Prolonged snow cover results in short growing seasons, which encourages the dominance of short herbs and graminoids, and prevents tall shrubs increasing in abundance. Longer growing seasons may reduce the habitat suitability for many characteristic plant species and lead to encroachment of adjacent ecosystem types.

Data availability and quality

Time-series data on phenology in *Snowpatch Herbfield* are not available. Hence, we cannot assess this indicator.

Criterion E: Quantitative analysis of probability of collapse

No stochastic models of *Snowpatch Herbfield* are available and there are presently insufficient data to reliably inform simulations of graminoid or shrub encroachment or change in snow melt date. Therefore, the risk status is **Data Deficient** under criterion E

Cushion Moorland

Authors

David Keith, Jamie Kirkpatrick, Jayne Balmer, Jessica Rowland

Reviewers

Chloe Sato, Emily Nicholson

Biome

T6 Polar-alpine

Functional group

T6.4 Temperate alpine grasslands and shrublands

IUCN status

Vulnerable

Cushion moorland. Mount Field National Park, 2007. Photo: Tim Rudman

Assessment Summary

Cushion Moorland (also known as Bolster Heath) is dominated by compact cushion plant growth forms and is found in areas of prolonged snow lie, poor drainage and/or extreme wind exposure within the alpine zone of Tasmania. It is threatened by climatic warming, which promotes increased frequency of heat waves and drought that cause cushion dieback, and invasion of the vegetation and cushion surface by non-cushion graminoids and shrubs, reducing cushion cover; together these processes also make the vegetation more flammable and increase the vulnerability of *Cushion Moorland* to fire. The ecosystem type has been assessed as **Vulnerable** based on restricted distribution (measured by Extent of Occurrence) in combination with threats likely to lead to continuing declines and a small number of threat-defined locations (sub-criterion B1), and a restricted distribution (small number of threat-defined locations) and fire-related threats that may cause the ecosystem type to collapse or become critically endangered within a very short period (sub-criterion B3) (Table 25).

Table 25. Summary of the Red List of Ecosystems assessment of the Australian *Cushion Moorland*. Category ranges in parentheses reflect uncertainty in assessment under the corresponding criteria.

Criteria	**A**	**B**	**C**	**D**	**E**	**Overall**
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	VU	DD	DD	DD	VU
Sub-criterion 2 *A, C, D: 50-year period including present & future* *B: AOO*	DD	LC	DD	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	LC	VU	DD	DD		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria

Ecosystem Description

Ecosystem Classification

In the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022a), this sub-global ecosystem type (Level 6) belongs to Ecosystem Functional Group *T6.4 Temperate alpine grasslands and shrublands* within the *Polar/alpine (Cryogenic) biome*.

Cushion Moorland assessed here is equivalent to Bolster Heath in (Venn et al. 2017) and has been described in Costin et al. (2000) and Kirkpatrick (1983, 1997). For the purposes of this assessment, areas that contain some cushion plant growth forms occurring in close association with *Snowpatch Herbfield* on mainland Australia (e.g., Kosciuszko National Park) were classified as Snowpatch ecosystems. Similarly, areas of *Fjaeldmark* that with cushion plant growth forms (e.g., *Colobanthus pulvinatus*) were mapped and classified as *Fjaeldmark* ecosystems (Kirkpatrick & Marsden-Smedley 2015; Parry et al. 2016). These areas are not included as part of the *Cushion Moorland* ecosystem type, and are not included or further discussed in this assessment, and *Cushion Moorland* is therefore endemic to Tasmania.

In Tasmania, this ecosystem type is described and mapped as *Cushion Moorland* (HCM) (Kitchener & Harris 2013; DPIPWE 2020). Specht (1970) accommodates this unit within either herbland or low shrubland categories depending on whether the dominant cushion species is woody or not. Under the Australian National Vegetation Information System, *Cushion Moorland* is classified as 'Low Heathland' and mapped as part of the major group 'Heathland' (NVIS Technical Working Group 2017)

Although not explicitly listed under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), *Cushion Moorland* occurs within National Heritage areas and is therefore considered a 'Matter of National Environmental Significance' under the EPBC Act.

Distinction from similar ecosystem types

Cushion Moorland shares some similarities with *Alpine Sphagnum Bog and Associated Fen* and *Tasmanian Alpine Sedgeland* in plant composition. However, *Cushion Moorland* is dominated by cushion plant growth forms, whereas *Alpine Sphagnum Bog and Associated Fen* have permanently waterlogged soils and are dominated by hummock-forming *Sphagnum* moss or wet heath shrubs, and *Tasmanian Alpine Sedgeland* is dominated by hard-leaved monocotyledonous plants that form mats or tussocks. *Cushion Moorland* is typically found in a matrix of *Tasmanian Alpine Heath* and *Tasmanian Alpine Sedgeland*. *Cushion Moorland* and *Tasmanian Alpine Sedgeland* are only found in Tasmania, whereas *Alpine Sphagnum Bog and Associated Fen* also extends to mainland Australia

Distribution

Cushion Moorland is located between 145.58° to 147.73° longitude and between -41.53° and -43.52° latitude, occurring in highland areas of Tasmania within four Tasmanian Bioregions: Ben Lomond, Central Highlands, Southern Ranges, and West (Kitchener & Harris 2013). In total, *Cushion Moorland* is currently mapped over an area of about 26.42 km^2 (Kitchener & Harris 2013; DPIPWE 2020). However, this area estimate has a large uncertainty due to the difficulty in mapping the distribution of small, widely dispersed patches of *Cushion Moorland*. There is no map of historical distribution or recent losses of *Cushion Moorland.* Spatial products used to create the present-day distribution map in this assessment (Figure 55) represent the most extensive and/or accurate mapping available within Tasmania for the defined bounds of *Cushion Moorland* at the time of assessment (April 2021), as advised by government representatives involved in the assessment process.

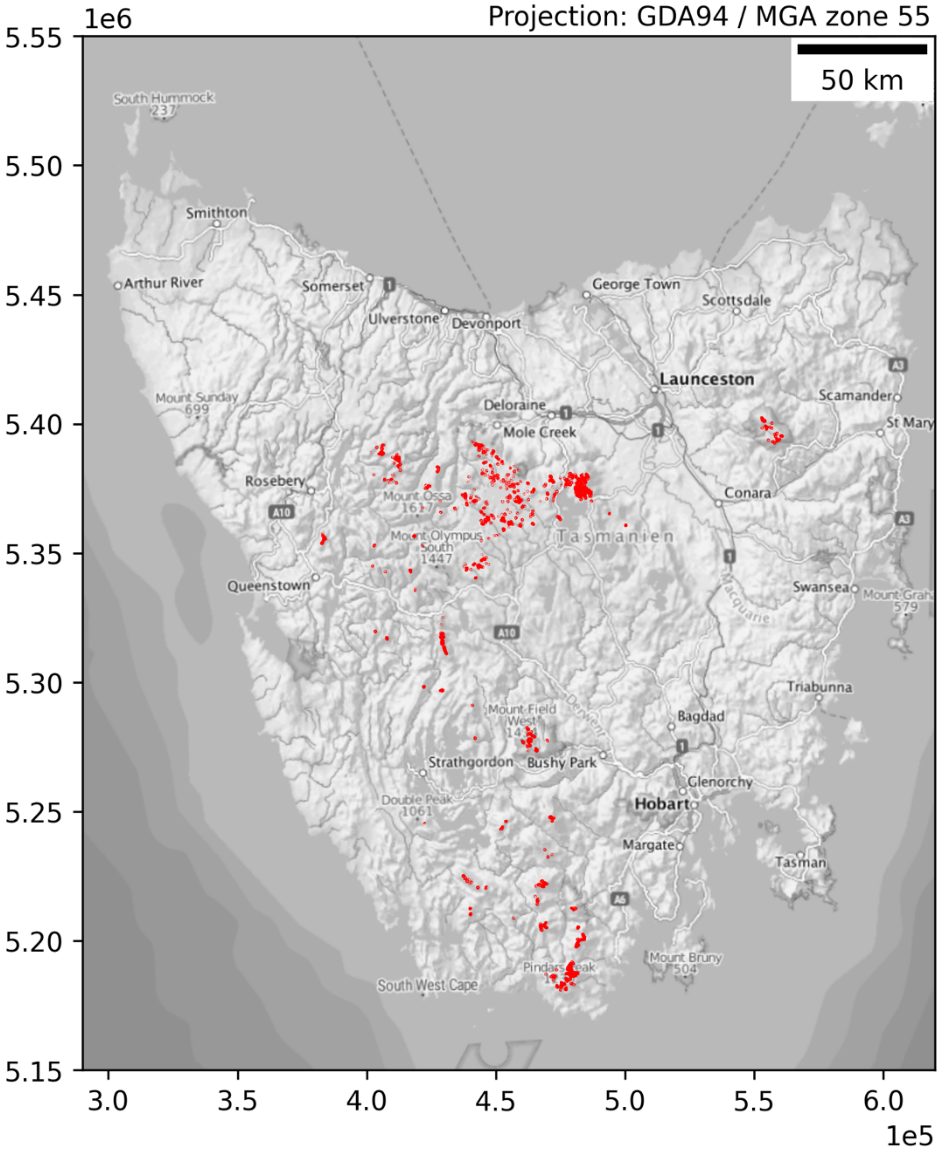


Figure 55. Distribution of *Cushion Moorland* (red) across Tasmania.

Abiotic environment

Most *Cushion Moorland* occurs in high rainfall areas where soils are constantly moist and typically poorly drained, most often in the flat bottoms of shallow headwater valleys (Kitchener & Harris 2013; Venn et al. 2017). Vegetation of *Cushion Moorland* can tolerate areas exposed to strong, ice-laden winds, and areas with persistent snow-lie. The ecosystem type occurs primarily above 900 m elevation and more than half is mapped above 1100 m in altitude. It is extremely rare or absent below 800 m (Kitchener & Harris 2013; DPIPWE 2020).

Characteristic native biota

Cushion Moorland is defined by vegetation in which cushion growth forms make up a greater ground cover than any other plant life-forms present (shrubs, herbs, graminoids or grasses; Figure 56), and bare ground is < 50%. Cushions are defined as plants in which the foliage forms a compact surface (Venn et al. 2017). Typically, areas meeting this definition form small, scattered patches within a matrix of other types of Tasmanian alpine vegetation dominated by heaths, sedges, rushes and herbs (Kitchener & Harris 2013).

Figure 56. Cushion plants regenerating vegetatively two years after fire amongst rapidly growing flammable *Empodisma minus* and sedges. Central Plateau, Tasmania. Photo: Jamie Kirkpatrick.

The cushion plants that dominate or are common and widespread in *Cushion Moorland* are Tasmanian endemic species, including *Abrotanella forsteroides*, *Donatia novae-zelandiae, Dracophyllum minimum, Ewartia meredithiae, Pterygopappus lawrencei, Oreobolus oligocephalus* and the non-endemic graminoid *Oreobolus pumilio subsp. pumilio* (Kirkpatrick 1997). Plant species that are less abundant or widespread but are generally associated with this ecosystem type include the cushion species *Phyllachne colensoi, Carpha rodwayi, Schizacme archeri, Colobanthus pulvinatus, Veronica ciliolata,* and *Prasophyllum alpinum* (Kirkpatrick 1997). Cushion epiphytes including *Euphrasia hookeri, Euphrasia gibbsiae subsp. pulvinestris, Plantago gunnii* and *Sprengelia minima* also occur in *Cushion Moorland* (Kirkpatrick 1997).

Fauna assemblages of this ecosystem type are poorly studied. At least one cushion moth, *Nemotyla oribates* (a species endemic to Tasmania), was found to be dependent on cushion plants. The larvae of this moth feed and tunnel in some cushion species (Nielsen et al. 1992; Edwards & McQuillan 1998). The cushions also provide a dry surface for resting mammals (*Vombatus ursinus* common wombat, *Macropus rufogriseus* Bennet's wallaby, and *Wallabia bicolor* swamp wallaby) and a mating substrate for many invertebrates (Kirkpatrick et al. 1993). Several plants in this ecosystem type are dependent on pollination by invertebrates (Kirkpatrick et al. 1993).

Key components, processes, and interactions

Growth rates and soil organic matter

Growth rates of cushion plants are slow, but their compact growth forms equip them to survive very cold temperatures, frosts and freezing winds, giving them an advantage over plants with other growth forms. *Cushion Moorland* that occurs in low gradient valley flats contribute to the accumulation of organic matter in soils. This leads to the blocking and braiding of streams and slows water movement downhill through the valley. In turn, slowed water movement generates positive feedback processes, enhancing the growth rates of cushions at the expense of graminoid and shrub growth. Enhanced cushion growth further accumulates organic matter in the soil and increases the microtopographic heterogeneity of the ground surface. Peat pool complexes may develop in localised areas, for example at Newdegate Pass, Mt Field National Park (Kirkpatrick & Gibson 1984). Conversely, fire, erosion and a drying climate may reduce or remove organic content in the soil, promoting an increased rate of water flow or drainage through these valleys to the detriment of *Cushion Moorland* (Figure 57; Figure 58).

Fire

The compact growth forms of cushion plants are non-flammable and relatively fire-resistant, even though co-occurring graminoids and shrubs are much more flammable and will carry fires through the ecosystem type when conditions are dry enough. Although the compact fuel structure of cushion plants will rarely propagate fire, the live foliage may be scorched by heat generated by combustion of nearby graminoids and shrubs. Some cushion species, such as *Dracophyllum minimum*, are fire sensitive and the scorching effects of fire adjacent to the plants can kill them. In other species, such as the widespread *Abrotanella forsteroides*, individuals that are only partially scorched can survive, regrow from well-insulated buds buried deeply in the cushion, and eventually replace the scorched foliage. Some individuals may be killed, however, if fire manages to enter the peat below the cushion surface resulting in the entire cushion smouldering. This is more common where sedges and shrubs have colonised the cushion surface or around the cushion edge and act as wicks for flame to burn into the cushion. If scorching damage is minimal, fire may advantage resprouting cushion species by reducing competition from fire-sensitive alpine shrubs within the community (Harding & Kirkpatrick 2018).

Figure 57. Cushion Moorland burnt in January 2016 (3 years prior to photograph) showing partial scorch and survival of *Abrotanella forsteroides* cushions (middle ground), while a few have been completely consumed by slow smouldering and their remains have been colonised by mosses (foreground). Near Lake Mackenzie, Central Plateau, Tasmania. Photo: David Keith.

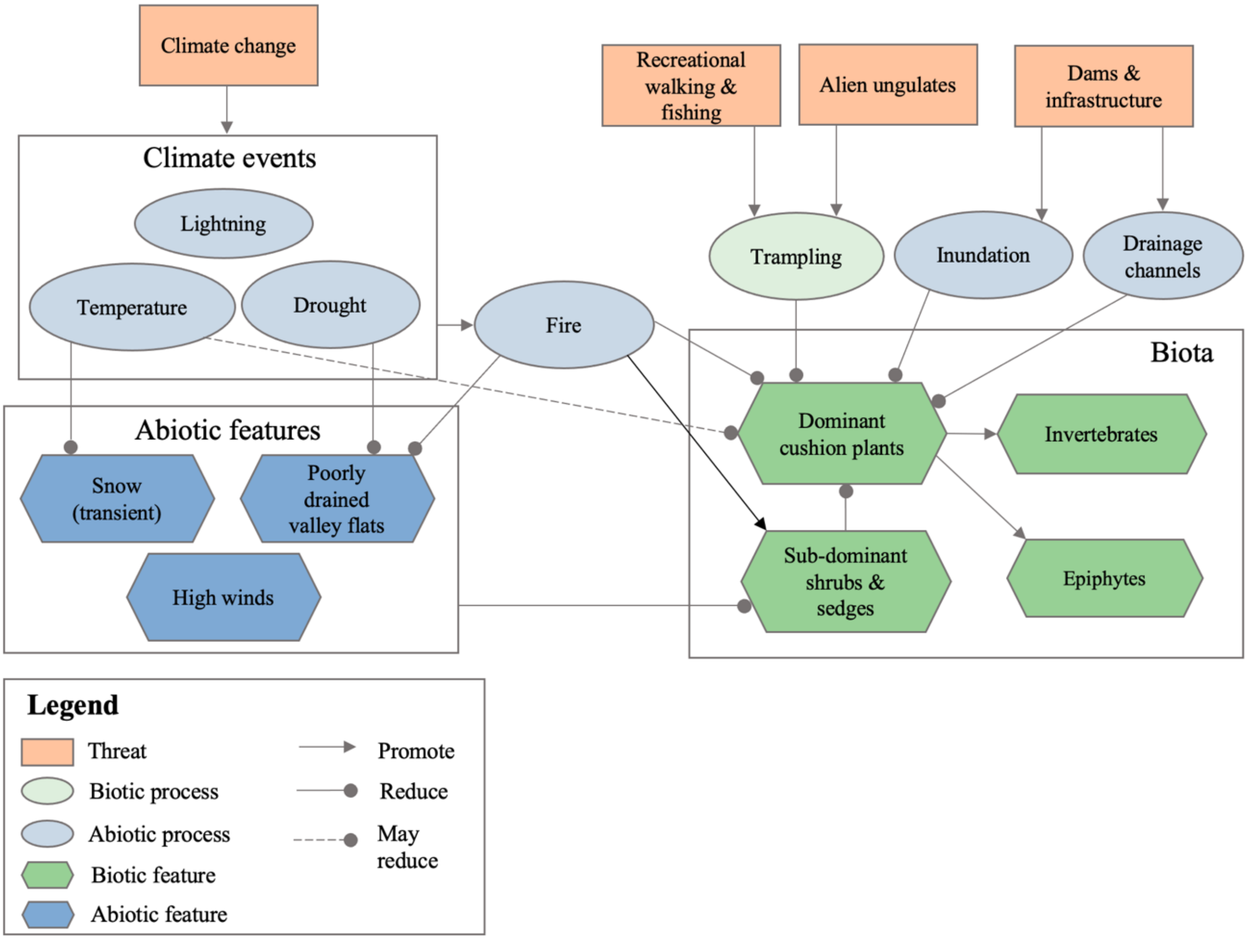


Figure 58. Conceptual diagram illustrating relationships between, and threats to, biotic and abiotic components and processes in *Cushion Moorland*.

Major threats

There is evidence from a few sites that the keystone cushion plant species in this ecosystem type may be lost or damaged through several threatening processes such as fire, inundation by artificial water storages, changed drainage, trampling, road or track development and competitive exclusion by native shrubs and/or graminoids due to successional processes (Gibson 1984; Kirkpatrick & Dickinson 1984; DPIPWE 2016b; Kirkpatrick et al. 2017).

Developments

Dams and infrastructure developments are key threats for this ecosystem type (Figure 58). Inundation by artificial lakes and dams for hydro-electricity generation is a threat to *Cushion Moorlands*. The ecosystem type can become permanently inundated beneath stored water and thus irreparably destroyed. Losses from inundation are restricted to part of the area of valley bottom flats marginal to original lake levels around Lake Mackenzie, Lake Augusta and Great Lake. These losses occurred pre-1970 and are poorly documented. Future expansion of hydro-power infrastructure may threaten additional areas of *Cushion Moorland.*

Soil drainage changes are associated with the construction of water races to maximise water capture within hydro-electric impoundments for use in power generation. This has resulted in deceased surface inflow and ground water to some *Cushion Moorlands* downslope (e.g., Ritters Plains below Lake Mackenzie, (DPIPWE 2016b)). Likewise, roads and drainage culverts may have impacted some local areas in proximity to such infrastructure. Overall, this is likely to cause increased frequency of drought conditions leading to dieback within the cushions, increased growth and engulfment of cushions by shrubs and graminoids, and increased ecosystem flammability. The condition of cushion surface is likely to decline enabling invasion by shrubs and graminoids that may act as wicks for ingress of fires.

Climate change

Climate change poses a large threat to *Cushion Moorland* from climatic stresses such as increased temperature extremes, reduced snow cover, prolonged droughts, and accelerated evapotranspiration, that may lead to dieback in cushions and increased competition and invasion by shrubs and graminoids at the expense of cushions . Pancotto et al. (2021) found that experimental passive surface warming of 0.4-0.7◦C reduced photosynthetic activity of cushion plants in the Andes, with carbon dioxide sequestration reduced by 55-85% relative to untreated controls over the main growing season. Conversely, warming is likely to promote faster growth rates of shrubs and graminoids (Dolezal et al. 2020) at the expense of cushion plant health. Severe dieback events in response to climate stress have been reported for the cushion plant, *Azorella macquariensis* on Macquarie Island (Bergstrom et al. 2015). More limited examples of dieback and invasion by shrubs and graminoids on and between the cushion surfaces have been observed by the authors (JB, JBK) in some parts of the Central Highlands (Figure 58), which is likely due to recorded reductions in summer rainfall, greater radiation, and more frequent extreme heat events (J. Balmer, pers. comm). The faster growth by shrubs and graminoids in warmer conditions allows cushions to be outcompeted and engulfed by these plants and increases susceptibility of cushions to fire.

Fire

Drier conditions and droughts associated with climate change and an increase in dry lightning strikes have led to an increase in the prevalence of fires in the ecosystem type (DPIPWE 2016b). The invasion of shrubs and graminoids increase the flammability of the ecosystem type and enables the ingress of fire into the cushion plants (Natural Values Conservation Branch 2017). Partial combustion of cushions can reduce their size initially. Recovery from low severity fires can be quite rapid but complete combustion causes mortality (Figure 59) (Harding & Kirkpatrick 2018). Loss of cushions allows graminoids and shrubs to further expand their dominance at the expense of the cushions and further promoting flammability of the site. Recruitment of new cushions is slow due to limited dispersal capacity and slow growth rates. At least 138 ha of *Cushion Moorland* have been burnt over the last 50 years (Tasmanian Government 2022), which have degraded but not caused complete loss of *Cushion Moorland*. Severe summer fires are predicted to increase over next 50 years.

Figure 59. Resprouting of some parts of the scorched cushion surface of *Abrotanella forsteroides* one year after fire in vicinity of Lake Mackenzie, Central Highlands. Note also the remains of fire killed shrubs emergent from dead cushion surface and the extensive resprouting of the graminoid, *Empodisma minus,* across the live and dead cushion surface. Photo: Micah Visoiu.

Trampling

Trampling and associated damage (e.g., soil erosion) by introduced ungulates, track development and off-track walking cause the breakup and drying of cushions, sometimes killing them (Gibson 1984). Trampling may also contribute to breaches of vegetative barriers to surface water flow, altering local structure and dynamics of the ecosystem, including peat ponds (Kirkpatrick 1997). Trampling by introduced ungulates occurred since 1850 (sheep) but was limited to private land. Disturbance by deer and goats are increasing at present in reserves and private land, and limited grazing by domestic livestock is likely to continue on private land.

IUCN Stresses Classification

1.2 – Ecosystem degradation
1.3 – Indirect ecosystem effects

IUCN Threats Classification

1.3 – Tourism & Recreation Areas
2.3.2 – Small-holder grazing, ranching or farming
6.1 – Recreational activities
7.2.10 – Large dams (hydro-electricity)
7.1.1 – Increase in fire frequency/intensity
8.1 – Invasive non-native/alien species/diseases (deer, goats)

11.2 – Droughts
11.3 – Temperature extremes

Ecosystem collapse

Cushion Moorland collapses when the characteristic cushion plants lose dominance. This may occur with a high frequency of high temperatures, droughts or fires leading to desiccation of the cushions. Consequently, bare ground, graminoids or shrubs may become dominant and transition the ecosystem type into *Tasmanian Alpine Heath*, *Alpine Sphagnum Bog and Associated Fen* or *Tasmanian Alpine Sedgeland*.

Cushion Moorland collapses when any of the following occur:

- *Area*: The mapped distribution declines to zero (100% loss) (Criteria A and B), or;
- *Cushion plant cover*: cushion plants become subordinate to graminoids or shrubs, specifically when cushion plants make up a smaller ground cover than any other plant life-form group (Criterion D), and/or;
- *Bare ground*: when bare ground is > 50 % cover (Criterion D).

Risk Assessment

Criterion A: Reduction in distribution

Summary

There are likely to be ongoing reductions in the distribution of *Cushion Moorland* due to fires, ingress by faster growing graminoids and shrubs associated with climate warming, and localised infrastructure development. These changes in distribution have not been quantified but appear very unlikely to have exceeded a 30% reduction in the past 50 years or since European invasion. No projections of future changes in distribution have been made. Hence, the status of this ecosystem type is likely to be **Least Concern** based on sub-criteria A1 and A3, while sub-criterion A2 is **Data Deficient**.

Criterion B: Restricted distribution

Summary

The status of *Cushion Moorland* is assessed as **Vulnerable** under sub-criterion B1 and B3, and **Least Concern** under sub-criterion B2.

Methods

Under criterion B, the extent of occurrence (EOO; B1) and area of occupancy (AOO; B2) of *Cushion Moorland* were determined using existing map products from Tasmania (Kitchener & Harris 2013; DPIPWE 2020). A complete description of the data and methods used to create the current distribution map for this ecosystem type is provided in main methods section.

The number of threat-defined locations was based on climate change and fire as these are the most important (and related) threats to *Cushion Moorland*.

Assessment outcome

Sub-criterion B1

A minimum convex polygon enclosing all mapped occurrences of *Cushion Moorland* has an area of 23,336.01 km^2. Based on projected climate warming throughout the distribution, a continuing decline in the distribution and function of the ecosystem type is likely due to the resulting ingress of graminoids and shrubs (see criterion D), and increased incidence of alpine fires (see criterion C). Climate change and fires are likely to affect patches semi-independently due to effects of local refuges, but there are likely to be ten or less threat-defined locations. *Cushion Moorland* therefore meets sub-criterion B1 for listing as **Vulnerable.**

Sub-criterion B2

The distribution of *Cushion Moorland* intersects a minimum of 73 10×10 km grid cells (Figure 60). This does not meet the threshold value for Vulnerable, thus the ecosystem type is listed under sub-criterion B2 as **Least Concern**.

Sub-criterion B3

The most serious plausible threats to *Cushion Moorland* are related to climate warming, including fires, extreme temperatures, and ingress of non-cushion plant life forms. Climate change and fires are likely to affect patches semi-independently due to effects of local refuges, but there are plausibly ten or less threat-defined locations. These climate-related threats could cause the ecosystem type to collapse or become Critically Endangered within a very short period of time (c. 20 years). It therefore meets the requirements for **Vulnerable** status under sub-criterion B3.

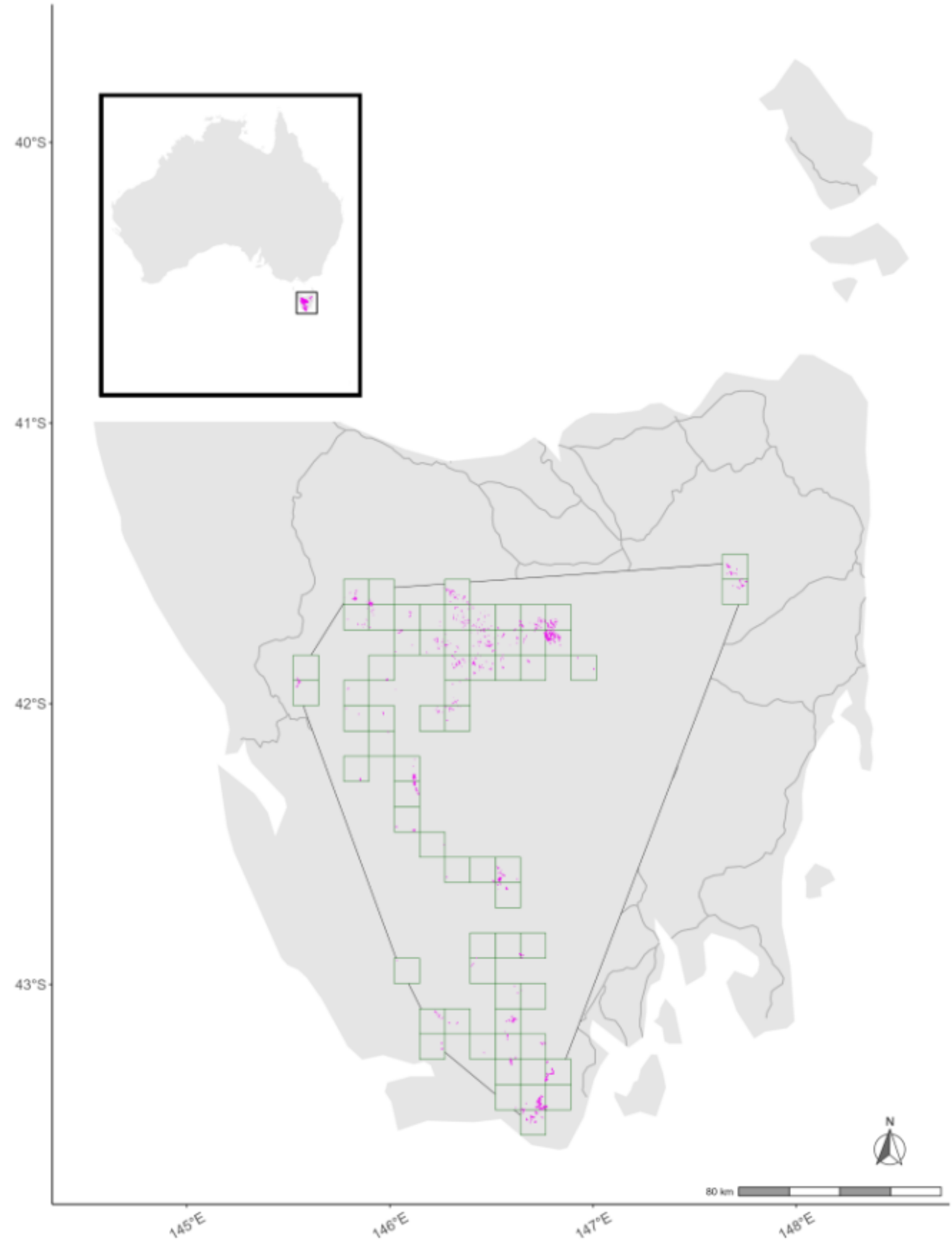


Figure 60. Map of *Cushion Moorland* (magenta polygons) showing EOO (black polygon) and AOO where the 1% rule was not applied (green squares).

Criterion C: Environmental degradation

Summary

Cushion Moorland is susceptible to environmental degradation related to climate change, as the dominant cushion plants are likely to be sensitive to increased frequency of high temperatures, droughts, and fires. However, the temperature and desiccation conditions associated with cushion dieback have not been quantified, nor has the response of cushions to recurring fires. The ecosystem type is also characterised by low cover of bare ground (< 50%), but there are insufficient data to assess this indicator. Hence, there are insufficient data to assess this process, and the status of the ecosystem type is **Data Deficient** under criterion C.

Identification of abiotic indicator

We examined the relevance and data availability/quality for one abiotic indicator to assess the risk of collapse from disruption of biotic processes and/or interactions:

- *Bare ground cover*: a measure of extent of bare ground and thus lack of vegetation.

Indicator: Bare ground cover

Relevance to ecosystem function

Cushion Moorland is characterised by vegetation that form a ground cover (Kirkpatrick 1997), with bare ground typically covering < 50% of the ecosystem type. The increase of bare ground cover, and thus the loss of the characteristic cushion plants cover, would lead to the loss of the ecosystem type and transition to *Tasmanian Alpine Heath*, *Alpine Sphagnum Bog and Associated Fen* or *Tasmanian Alpine Sedgeland.*

Selection of collapse threshold

We consider the ecosystem type to collapse when bare ground cover is > 50%.

Data availability and quality

There are insufficient data to assess this indicator.

Criterion D: Disruption of biotic processes and/or interactions

Summary

Biotic processes within *Cushion Moorland* hinge on the abundance of cushion plants as keystone species. No time series of field observations are available to assess relevant trends in the abundance of cushion plants, however such an analysis could be plausible with high-resolution remote sensing data. Currently, the ecosystem type is assessed as **Data Deficient** under criterion D.

Identification of biotic indicator

We examined the relevance and data availability/quality for two biotic indicators to assess the risk of collapse from disruption of biotic processes and/or interactions:

- *Cushion plant cover*: a measure of the cover of the characteristic cushion plant species.

Indicator: Cushion plant cover

Relevance to ecosystem function

Cushion Moorland is characterised by cushion plants that form a ground cover (Kirkpatrick 1997). Common species include *Abrotanella forsteroides, Donatia novae-zelandiae, Dracophyllum minimum, Phyllachne collensoi, Ewartia meredithiae, Pterygopappus lawrencei, Oreobolus oligocephalus.* The loss of these cushion plants would lead to the loss of the ecosystem type and transition to *Tasmanian Alpine Heath*, *Alpine Sphagnum Bog and Associated Fen* or *Tasmanian Alpine Sedgeland*.

Selection of collapse threshold

We consider the ecosystem type to collapse when the proportion of cushion plant cover is lower than any other plant life-form group.

Data availability and quality

There are insufficient data to assess this indicator.

Criterion E: Quantitative analysis of probability of collapse

Summary

Insufficient data are available to estimate the probability that *Cushion Moorland* will collapse within the next 50 to 100 years. Hence the ecosystem type is **Data Deficient** under criterion E.

Cushion moorland in southwest Tasmania. Photo Louise Gilfedder

Tasmanian Alpine Heath

Authors

David Keith, Jamie Kirkpatrick, Jayne Balmer, Jessica Rowland, Ella Plumanns Pouton

Reviewers

Clare Vernon, Tracey Regan

Biome

T6 Polar-alpine

Functional group

T6.4 Temperate alpine grasslands and shrublands

IUCN status

Near Threatened (Least Concern-Near Threatened)

Tasmanian Alpine Heath, Hartz Mountains, December 2024. Photo: Jayne Balmer

Assessment Summary

Tasmanian Alpine Heath is dominated by a highly diverse assemblage of largely Tasmanian endemic scleromorphic shrubs adapted to winter exposure to the extreme cold winds and intermittent snow. It is the most widespread alpine community in Tasmania (890 km^2), occurring on a wide variety of substrates and topographic situations. Most of the area of *Tasmanian Alpine Heath* is potentially suitable for occupancy by coniferous or deciduous heath, and in some areas these elements may been removed by past fires. Consequently, the community has expanded since European colonisation of Tasmania and is likely to continue to expand with climate change. Climate change may promote transformation of coniferous and deciduous heath with *Tasmanian Alpine Heath* as scrub and forest move upslope. Some minor past and future losses relate to alpine development for hydro-electric power and tourism as well as accelerated erosion resulting from the combination of fire and grazing.

Tasmanian Alpine Heath does not meet any of the criteria for listing in a threatened category. Its distribution is moderately restricted (based on Extent of Occurrence), but current threats are unlikely to cause continuing declines. Therefore, it was assessed as **Near Threatened** (**Least Concern to Near Threatened**) (Table 26).

Table 26. Summary of the Red List of Ecosystems assessment of the *Tasmanian Alpine Heath*. Category ranges in parentheses reflect uncertainty in assessment under the corresponding criteria.

Criteria	**A**	**B**	**C**	**D**	**E**	**Overall**
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	NT (LC-NT)	LC	DD	DD	NT (LC-NT)
Sub-criterion 2 *A, C, D: 50-year period including present & future* *B: AOO*	LC	LC	DD	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	LC	NT	DD	DD		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria

Ecosystem Description

Ecosystem Classification

In the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022), *Tasmanian Alpine Heath* is a global ecosystem type (Level 5) belonging to Ecosystem Functional Group *T6.4 Temperate alpine grasslands and shrublands*, within the *T6. Polar-alpine (cryogenic)* biome.

Within Australia, *Tasmanian Alpine Heath* was described as alpine heath by Kirkpatrick (1983, 1997), and has been mapped as Eastern alpine Heathland (HHE) and Western alpine heathland (HHW) (Kitchener & Harris 2013; DPIPWE 2020). The properties and dynamics of *Tasmanian Alpine Heath* contribute to the outstanding natural significance of the Western Tasmanian Wilderness World Heritage Area which is protected under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC).

Distinction from similar ecosystem types

Tasmanian Alpine Heath (Tasmania only) is similar in structure to *Coniferous Heath* (mainland and Tasmania) and *Alpine –sub-alpine Closed Heath* (mainland only). *Tasmanian Alpine Heath* is dominated by a diverse assemblage of largely Tasmanian endemic scleromorphic shrubs exposed to strong winter winds, whereas *Coniferous Heath* is dominated by paleo-endemic coniferous shrubs, and *Alpine –sub-alpine Closed Heath* is defined by a closed canopy of fire-tolerant shrubs but lacks the Tasmanian endemic species.

Distribution

Tasmanian Alpine Heath is restricted to Ben Lomond, the Central plateau, the west coast ranges and the southern mountains of Tasmania, between 145.47° to 148.01° longitude and between -41.20° and -43.53° latitude. In total, *Tasmanian Alpine Heath* covers 893.79 km^2, occupying more than 60 alpine habitat islands (i.e., individual mountain peaks above the treeline) (Figure 61). Spatial products used to create the present-day distribution map in this assessment (Figure 61; DPIPWE, 2020) represent the most extensive and/or accurate mapping available within Tasmania for the defined bounds of *Tasmanian Alpine Heath* at the time of assessment (April 2021).

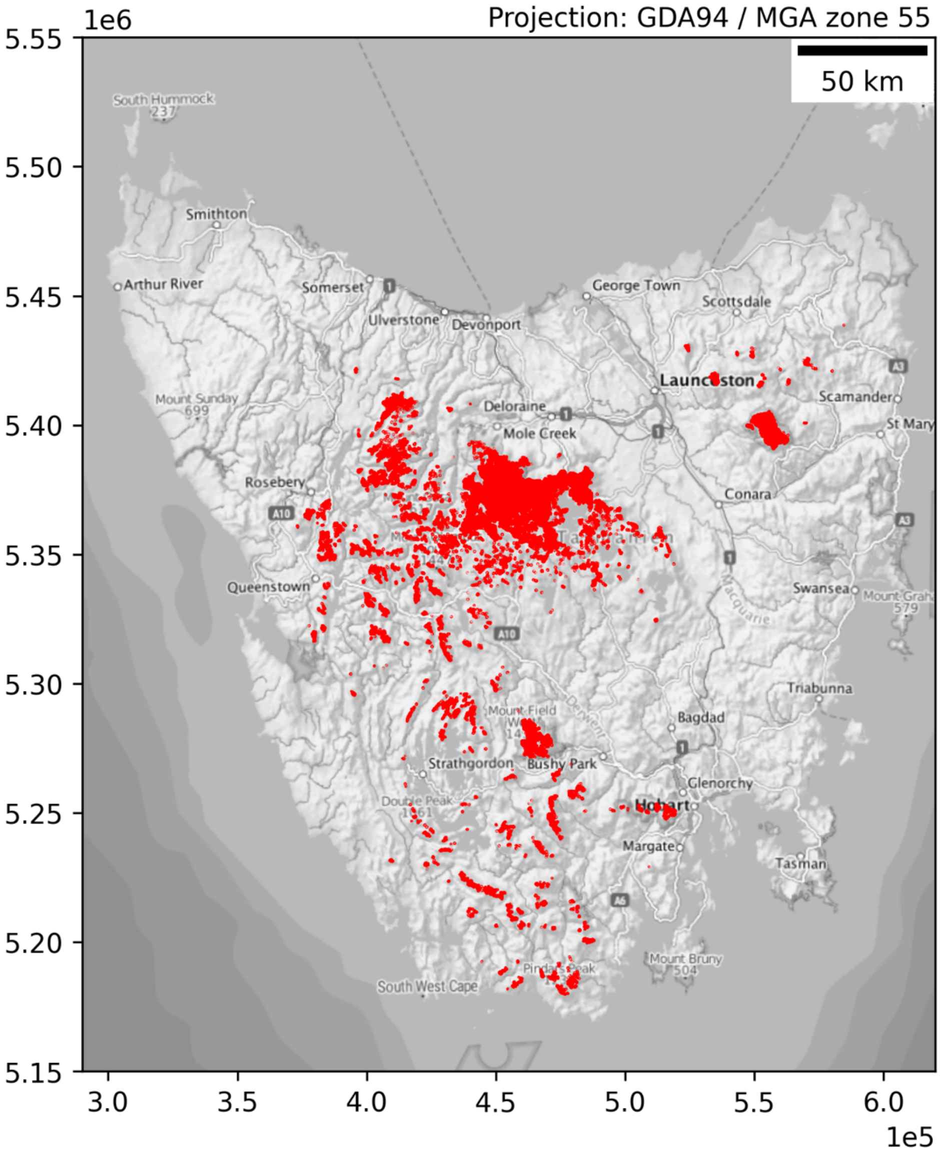


Figure 61. Distribution of the *Tasmanian Alpine Heath* (red) across Tasmania.

Abiotic environment

Tasmanian Alpine Heath occurs in a wide variety of environments, from fertile to infertile, rocky to shallow mineral soils, poorly-drained to well-drained and intermediate to high altitude (from 600-700 m to more than 1400 m above sea level) (Harris & Kitchener 2005; Kitchener & Harris 2013; Figure 62). All extents have in common an exposure of shrubs to fierce winter winds, as snow cover does not persist, and exposure to grazing by native mammals, which disperse to the highest elevations. They are characterised by very high rainfall, but occur within a broader precipitation envelope than other ecosystem types dominated by coniferous, deciduous and cushion plants.

Figure 62. Tasmanian Alpine Heath. Hartz Mountain, southern Tasmania. Photo: Jamie Kirkpatrick (2011).

Characteristic native biota

Tasmanian Alpine Heath is dominated by scleromorphic shrubs, which may form a closed or relatively open canopy, depending on the rockiness of the substrate and time since last fire. Grasses are relatively scarce, but scleromorphic sedges, rushes and forbs may make up an open ground layer amongst shrubs. Characteristic shrubs include *Orites acicularis* and *Richea acerosa*. Rockier sites support varied combinations of *Leptospermum rupestre, Richea sprengelioides, Ozothamnus rodwayi, Olearia pinifolia, Olearia ledifolia* and *Orites revoluta*. Other shrubs include *Montitega dealbata, Epacris serpyllifolia* and *Bellendena montana*. Burnt areas initially have a high cover of graminoids and are soon dominated by rapidly growing recruits *of Olearia algida, Ozothamnus hookeri* and more rarely *Ozothamnus ledifolius*, along with vegetatively regenerating *Orites revoluta, Richea acerosa* and *R. sprengelioides* (Kirkpatrick & Dickinson 1984; Kirkpatrick et al. 2002; Kirkpatrick & Bridle 2013). *Richea scoparia* and *Baeckea gunniana* occur on damper soils (Kirkpatrick 1997). Many of these species are endemic to Tasmania and different shrub assemblages occur in the eastern and western mountains (Kirkpatrick 1982, 1983). The main ground layer plants include the grass *Poa rodwayi*, sedges and rushes including *Carpha alpina* and *Luzula* spp., and forbs including *Celmisia asteliifolia*, *Euphrasia* spp. and *Acaena montana*.

The mammal fauna of Tasmanian Alpine Heath includes Antechinus swainsonii, Pseudomys fuscus, P. higginsi, Vombatus ursinus, Macropus rufogriseus, Wallabia bicolor, Tachvglossus aculeatus, Dasyurus·viverrinus and Sarcophiilus harrisii (Kirkpatrick et al. 1993). The avifauna includes raptors such as Aquila audax and Falco peregrinus, the raven Corvus tasmanicus, parrots such as

Calyptorhynchus funereus and Platycercus caledonicus, the thrush Zoothera dauma, Tasmanian thornbill Acanthiza ewingii, several species of robin Petroica spp., and the Tasmanian currawong Strepera fuliginosa. Distinctive reptiles include the endemic alpine skinks Niveoscincus orocryptus, N. greeni and N. microlepidotus. The invertebrate fauna is also distinctive with endemic katydids, lepidopterans, coleopterans, amphipods, lacewings, mites and spiders (Kirkpatrick et al. 1993)

Key components, processes, and interactions

Fire

The dominance of sclerophyll shrubs and mineral soils characteristic of this ecosystem type are maintained by periodic fires at multi-decadal intervals (Figure 63). Some of the shrub species survive fire and resprout from woody rootstocks, but most are obligate seeders where established plants are killed by fire and regeneration occurs from soil seedbanks. Post-fire regrowth is slow and may take 50-60 years to attain full structural recovery (Kirkpatrick et al. 2002). Fires promote germination of sclerophyll shrubs from soil seedbanks, but germination may also occur spontaneously during fire intervals and probably requires cold stratification of seeds (Venn et al., 2021). Higher frequency fires may eliminate slower growing shrubs and promote growth of graminoids, but slow fuel accumulation limits fire recurrence except under extreme fire weather conditions. Trees and other species of shrubs, abundant in warmer lowland environments, may outcompete the unique assemblage of alpine shrub species (Venn et al., 2014), but are limited by frosts and exposure to cold winds.

Browsing

Browsing mammals are present, but are unlikely to limit vegetation, except in the post-fire recruitment phase (Kirkpatrick et al. 2016); their populations in the alpine zone are limited by periodic snow (Green 2016). Skinks may be important in pollination of some species, along with dipterans (Johnson & Karen Johnson 2012).

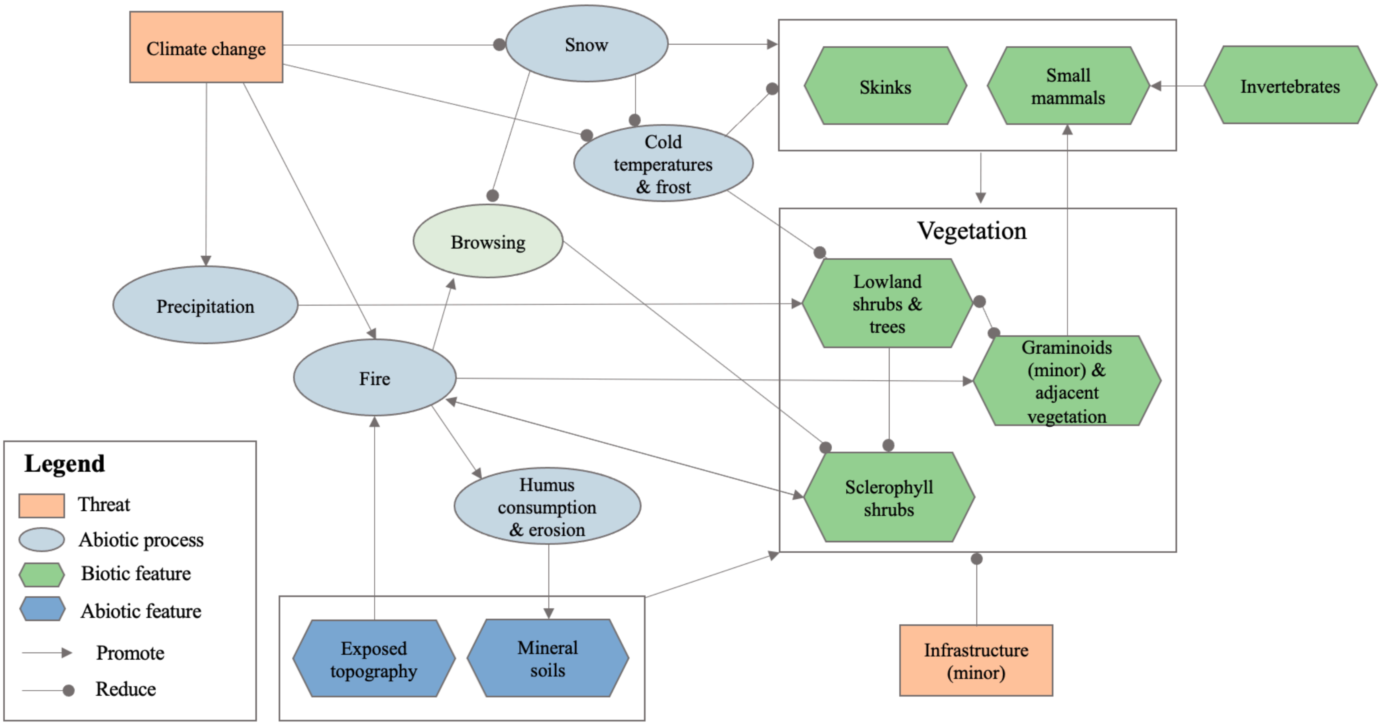


Figure 63. Conceptual diagram illustrating relationships between, and threats to, biotic and abiotic components and processes in *Tasmanian Alpine Heath*.

Major threats

The primary threats to *Tasmanian Alpine Heath* are related to frequent fire, climate change, and their interaction, and localised habitat loss due to infrastructure development. The effects of browsing by herbivores appear to be minimal but require further examination on the Central Plateau.

Fire

The current extent of *Tasmanian Alpine Heath* may be an outcome of past burning of coniferous and deciduous heath, which take centuries to recover their structure and composition after fire allowing alpine heath to replace it (Kirkpatrick et al. 2010). Burning of *Tasmanian Alpine Heath* is followed by succession back to the original state over 50-60 years (Kirkpatrick et al. 2002, 2010). This slow successional process could be interrupted by subsequent fire, resulting in declines or local extinctions of alpine shrub species, particularly obligate seeders reliant on seedbanks for postfire recovery. Invertebrates may also be susceptible to fires. Under typical conditions, fuel levels are too low to support recurrence of fire for several decades. However, climate change is projected to increase fire frequency and severity in the future (Abram et al. 2021; Canadell et al. 2021). A combination of fire and heavy grazing can induce sheet erosion which patchily eliminates the community, as on the Central Plateau, where recovery is slow to non-existent (Kirkpatrick & Bridle 2013). Fire impacts may be accentuated when fire events are followed by prolonged drought - a scenario that is expected to become more frequent with global warming.

Climate warming

A warming climate, especially in the east of Tasmania, is enabling rapid growth of shrubs and trees that might result in transformation of *Tasmanian Alpine Heath* to subalpine ecosystem types, such as *Alpine-subalpine Woodland and Forest* (DPIPWE 2010). There are early signs of this transition on Mt Rufus in an area burned several decades ago (Harrison-Day et al. 2016). Changes in precipitation are likely to shift species distributions within *Tasmanian Alpine Heath* (DPIPWE 2010), rather than change the distribution of the ecosystem type. Climate change is likely to exacerbate fire-related threats due to projections of increased frequency of extreme fire weather (Abram et al. 2021; Canadell et al. 2021).

Grazing

Fallow deer and rabbits graze the community on the Central Plateau. At present, rabbits have little or no influence on the vegetation (Kirkpatrick & Bridle 2013) and the effects of deer on Tasmanian Alpine Heath are unknown. They may eat *Astelia alpina*, which is otherwise untouched by mammals (Nichols & Kirkpatrick 2019). Browsing by native herbivores may become significant when their populations build up in the absence of prolonged snow cover in winter; wallabies suffer high mortality in snow (Green 2016).

Developments

Road and dam building has cleared localised areas of *Tasmanian Alpine Heath*. Proposed cable cars and associated development might remove more areas, as might some ecotourism development that has been encouraged through the State Government Expression of Interest process.

IUCN Stresses Classification

1.2 – Ecosystem degradation
1.3 – Indirect ecosystem effects

IUCN Threats Classification

1.3 – Tourism and recreation areas
4.1 – Roads & railroads
7.1.1 – Increase in fire frequency / intensity
7.2 – Dams & Water Management/Use
8.1 – Invasive Non-native/Alien Species
11.1 – Habitat shifting and alteration
11.2 – Droughts

Ecosystem collapse

Tasmanian Alpine Heath collapses when the characteristic heath species lose dominance, either in response to the development of a tree stratum with a cover of more than 5% or due to the cover of another life-form such as graminoids exceeding the cover of shrubs. *Changes in precipitation and warmer temperatures under climate change may enable the growth of shrubs and trees from adjacent ecosystems, triggering a shift to montane vegetation types* (DPIPWE 2010). *Tasmanian Alpine Heath* may become increasingly sensitive to frequent fire events, causing a transformation to *Tasmanian Alpine Sedgeland*.

Tasmanian Alpine Heath has collapsed in this assessment when any of the following occurs:

- *Area:* The mapped distribution declines to zero (100% loss) (Criteria A and B), or;
- *Fire intervals*: all patches have been burnt at an interval of < 20 years between successive fires within a 50-year period, causing major population reductions or elimination of sclerophyll shrubs (Criterion C).

Risk Assessment

Criterion A: Reduction in distribution

Summary

Since European invasion (c. 1800-1850), the area of this ecosystem type has increased due to the transformation of large areas of *Coniferous Heath* and the parts of the *Subalpine Paleoendemic Forest and Woodland* dominated by deciduous heath, *Nothofagus gunnii*, into *Tasmanian Alpine Heath* because of fire (Kirkpatrick et al. 2010). Loss of *Tasmanian Alpine Heath* caused by sheet erosion and infrastructure development are small in relation to this increase. It is likely that the area of *Tasmanian Alpine Heath* will increase in the next 50 years, as it has in the last 50 years and since European colonisation. The increase in the distribution of the ecosystem type over the past, present, and future therefore indicates a status of **Least Concern** under sub-criteria A1, A2 and A3.

Criterion B: Restricted distribution

Summary

Tasmanian Alpine Heath marginally fails to meet the criteria for listing in the Vulnerable category and therefore its status is assessed as **Near Threatened (Least Concern – Near Threatened)**. under sub-criterion B1 and **Near Threatened** under sub-criterion B3. However, its Area of Occupancy is substantially greater than the thresholds for threat categories and was therefore assessed as **Least Concern** under sub-criterion B2.

Methods

Under criterion B, the extent of occurrence (EOO; sub-criterion B1) and area of occupancy (AOO; sub-criterion B2) of *Tasmanian Alpine Heath* were determined using existing map products from Tasmania (Kitchener & Harris 2013; DPIPWE 2020). A complete description of the data and methods used to create the current distribution map for this ecosystem type is provided in the main methods.

The number of threat-defined locations was based on fire as this is the most important stochastic threat to *Tasmanian Alpine Heath*.

Assessment outcome

Sub-criterion B1

A minimum convex polygon enclosing all occurrences of *Tasmanian Alpine Heath* includes 34,548 km^2. However, the extent of *Tasmanian Alpine Heath* appears to be expanding at the expense of other heathland ecosystem types and there is no evidence that current threats are likely to cause continuing declines in extent or function of the ecosystem type in the foreseeable future. Consequently, none of the sub-criteria within criterion B1 are met, even though the estimated EOO is within the threshold for the Vulnerable category. The status of the ecosystem type under sub-criterion B1 is therefore **Near Threatened (Least Concern – Near Threatened)**.

Sub-criterion B2

The distribution of *Tasmanian Alpine Heath* intersects a minimum of 206 10×10 km grid cells (Figure 64). The ecosystem type therefore does not meet the Area of Occupancy (AOO) thresholds for any threatened category. Its status under sub-criterion B2 is therefore **Least Concern**.

Sub-criterion B3

The most serious plausible threat to *Tasmanian Alpine Heath* is fires recurring at decadal frequencies or in combination with drought. The entire distribution of *Tasmanian Alpine Heath* could plausibly burn in < 5 independent fire events within a short period of time, hence this ecosystem type may occur at ≤ 5 locations. However, this is unlikely to cause the ecosystem type to collapse or to become Critically Endangered within a short period of time (c. 20 years) because the likelihood of a second fire across all 5 locations within that timeframe is low. The status of the ecosystem type under sub-criterion B3 is therefore **Near Threatened**.

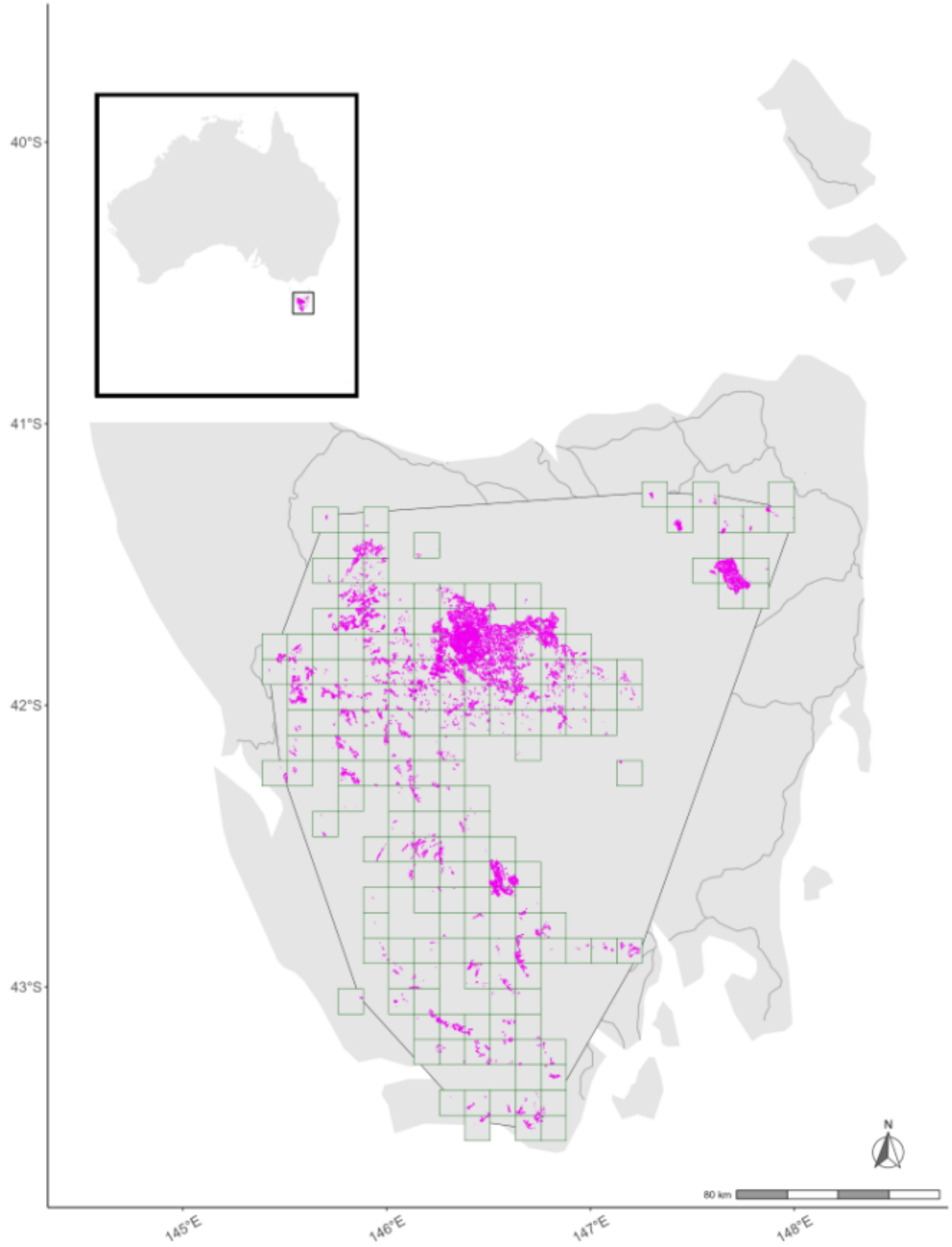


Figure 64. Map of *Tasmanian Alpine Heath (*magenta polygons) showing EOO (black polygon) and AOO where the 1% rule was not applied (green squares).

Criterion C: Environmental degradation

Summary

The status of *Tasmanian Alpine Heath* is assessed as **Least Concern** under sub-criteria C1. There are insufficient data to assess potential future and historical changes, and therefore the ecosystem type is assessed as **Data Deficient** for sub-criteria C2 and C3.

Identification of abiotic indicator

We examined the relevance and data availability/quality for one potential abiotic indicator to assess the risk of collapse from environmental degradation:

- *Fire frequency:* a direct measure of the frequency of fires in *Tasmanian Alpine Heath.*

Indicator: Fire frequency

Relevance to ecosystem function

Tasmanian Alpine Heath can slowly recover after fire; vegetation returns to the pre-fire state within 50-60 years (Kirkpatrick et al. 2002). Yet climate change is changing fire regimes in Australian alpine ecosystems (Abram et al. 2021; Canadell et al. 2021), increasing the frequency of fires due to more regular extreme fire weather, severe and longer droughts (Zylstra 2018; Dowdy et al. 2019), and ignitions from lightning (Styger et al. 2018). Changes in the frequency of fires may disrupt the slow recovery of the ecosystem type if intervals between fires become too short, resulting in species declines or local extinctions of characteristic shrubs as many endemic species are not fire tolerant (Kirkpatrick et al. 2010). For example, many plant species are obligate seeders and require a seedbank to recover post-fire, while resprouters my exhibit limited tolerance to recurrent fires as established plants and high mortality among juveniles that are essential to replace deaths of adults (Keith 1996). Therefore, *Tasmanian Alpine Heath* may become increasingly sensitive to frequent fire events, which may cause the ecosystem type to transform into another ecosystem type.

Data availability and quality

Recent change (C1) and Historical change (C3)

Historical records of bushfires and planned burns are available as shapefiles for Tasmania (Tasmanian Government 2022). The fire history dataset for C1 includes fires in the 1960/1961 and 2019/2020 fire seasons and for C3 spanning 1938/1938 to 2019/2020 fire seasons in regions above 600 m above sea level (Tasmanian Government 2022). Fire seasons are defined as fires that occur between July in one year and June in the next year.

Future change (C2)

There were insufficient data to assess future fire risk

Selection of collapse threshold

Based on slow recovery rates documented for *Tasmanian Alpine Heath* (Kirkpatrick et al. 2002), the ecosystem type was assumed to collapse when all patches have been burnt at an interval of < 20 years between successive fires in a 50-year period, causing major population reductions or elimination of sclerophyll shrubs. We translated this to a threshold of ≥ 2 fires within 50 years.

Selection of initial and present/future values

Recent change (C1)

To define the initial and present fire frequency, we analysed the fire history dataset from the Tasmanian state governments (Tasmanian Government 2022). We overlayed the fire history layers and extracted areas where the ecosystem type had been burnt. To generate a time-series, we calculated the number of times and area burnt for 1960-1980 (initial value) and 1970-2020 (current value). We used these values to calculate the mean annual probability of burning based on the spatially weighted mean area of the ecosystem type burnt at each frequency in each timeframe.

Historic change (C3)

We used the same approach as used in C1 to calculate initial and present values. In lieu of historical data, we assumed that the fire frequency in 1939-1949 was relatively representative of the fire frequency c. 1750.

Calculation of severity and extent

Recent change (C1)

Fires have burnt the ecosystem type in 28 fire seasons between 1970 and 2020, covering a total of 76.83 km^2, or 8.59% of the whole distribution. Leaving 91.41% of the ecosystem type unburnt within the past 50 years (Table 27). From 1970-2020, 2.6km^2 of the ecosystem type has burnt at or above the collapse threshold (two fires in 50 years), equating to 0.29% of the distribution (Table 27). The ecosystem type had a very low mean annual probability of burning between 1970-2020 (0.002). However, this increased from the 1939-1960 period, where no fires were recorded (Figure 65). The collapse threshold is an annual probability of 0.04 (2 in 50 years). This equates to a relative severity of 100 x ((0 – 0.002) / (0 – 0.04)) = 5%. Due to the minor increase in fire frequency the risk status is **Least Concern** under sub-criterion C1.

Future change (C2) and Historical change (C3)

Due to the lack of data, the ecosystem type is **Data Deficient** for sub-criteria C2 and C3.

Table 27. Percentage of the mainland extent of *Tasmanian Alpine Heath* burnt at each frequency and the spatially weighted mean annual probability of burning within each timeframe.

Fires	**1970-2020**	**1939-1960**
0	91.41	100
1	5.99	0
2	2.60	0
3	0.002	0
≥ 1/20 years		
Annual probability of burning	0.002	0.000

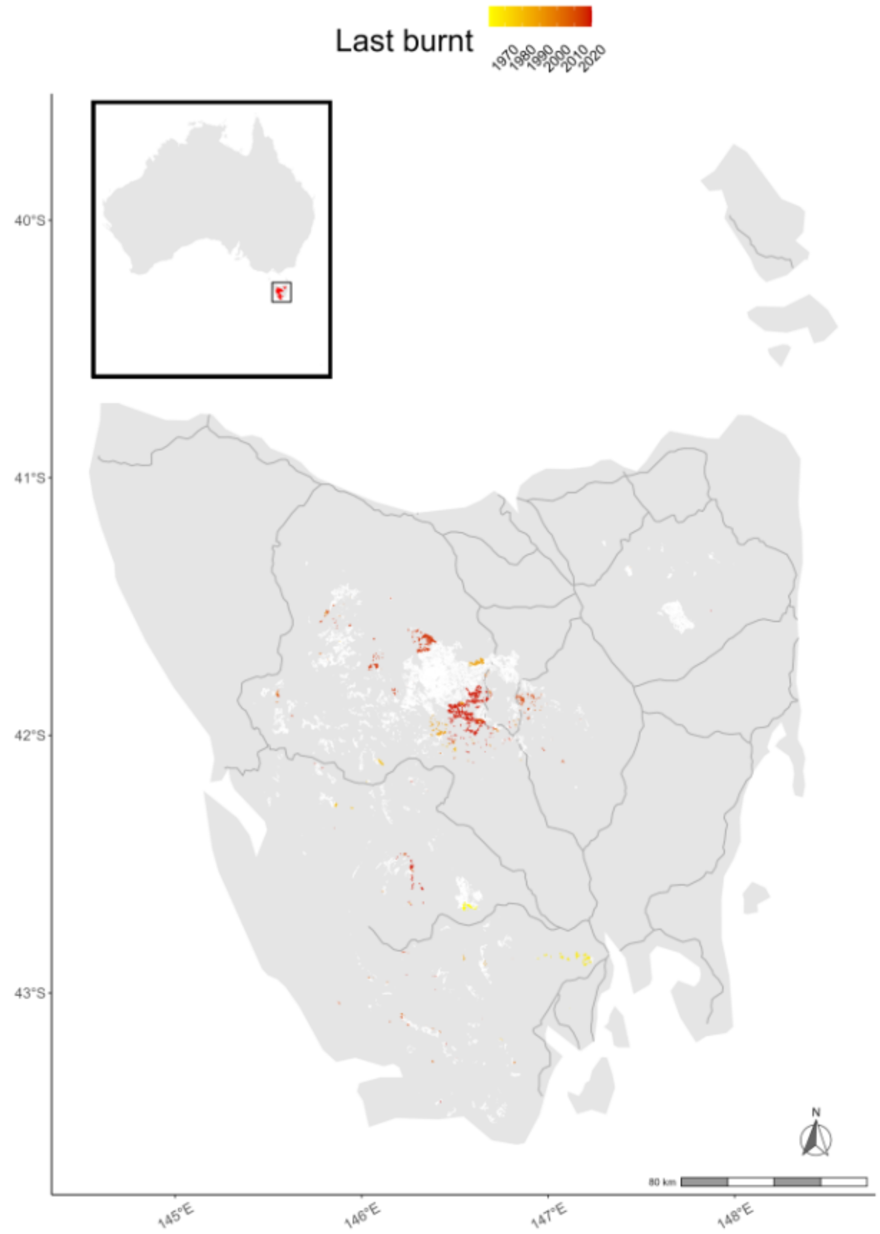


Figure 65. The time since the last fire in *Tasmanian Alpine Heath.* White areas show patches unburnt since 1939.

Criterion D: Disruption of biotic processes and/or interactions

Summary

Directional change in plant species composition related to climate change is the most relevant indicator of biotic disruption in *Tasmanian Alpine Heath*. Shifts in phenology may also alter the persistence of the characteristic plants and pollinator species. The available data are insufficient to draw a conclusion about rates of biotic disruption. Hence the status of the ecosystem type under criterion D is **Data Deficient.**

Identification of biotic indicator

We examined the relevance and data availability/quality for two biotic indicators, selected based on our conceptual model, to assess the risk of collapse from disruption of biotic processes and/or interactions:

- *Plant species composition*: a measure of the composition of the plant community
- *Phenology*: a measure of changes in the flowering of key plant species and presence of insect pollinators

Indicator: Plant species composition

Relevance to ecosystem function

Tasmanian Alpine Heath is characterised by scleromorphic shrubs (e.g., *Orites acicularis* and *Richea acerosa*) and some sedges, rushes and forbs (e.g, *Carpha alpina, Luzula spp., Celmisia asteliifolia, Euphrasia spp., Acaena montana*), whereas grasses (e.g., *Poa rodwayi*) are relatively scarce. The assemblages of shrubs can differ in the eastern and western range. Climate change is predicted to alter the plant species composition in alpine ecosystems as changes in precipitation and warmer temperatures enable the growth of shrubs and trees from adjacent ecosystems (DPIPWE 2010).

Data availability and quality

There is time-series data along the altitudinal gradient at Mt. Field which has demonstrated that between 1979and 2019 there was a minor upward shift in the distribution of several plant species (Minchin & Davies 2019ab, Minchin and Davies et al. 2022). However, insufficient data are currently available to assess indicator comprehensibly.

Indicator: Reproductive phenology

Relevance to ecosystem function

Reproductive phenology is the timing of events in species lifecycles and is influenced by seasonal and interannual climate variations. Climatic warming may alter reproductive phenologies of plants and animal pollinators (Visser & Both 2005; Jarrad et al. 2008; Hoffmann et al. 2010). As species can respond differently to climate warming (Jarrad et al. 2008), this may result in mismatches in the timing of flowering and pollination, there may be declines in plant reproductive rates and insect pollinator survival.

Data availability and quality

Insufficient data are available to assess this indicator as changes in phenology are currently undocumented for *Tasmanian Alpine Heath*.

Criterion E: Quantitative analysis of probability of collapse

Summary

There are no models or studies that quantify the risk of collapse of this ecosystem type over the next 50 or 100 years. Hence the status of the ecosystem type under criterion E is **Data Deficient.**

Tasmanian Alpine Heath Cradle Mountain area, Tasmania, October 2020. Photo: Jayne Balmer.

Tasmanian Alpine Sedgeland

Authors

David Keith, Jamie Kirkpatrick, Jayne Balmer, Jessica Rowland

Reviewers

Chloe Sato, Emily Nicholson

Biome

T6 Polar-alpine

Functional group

T6.4 Temperate alpine grasslands and shrublands

IUCN status

Near Threatened (Least Concern-Near Threatened)

Tasmanian Alpine Sedgeland on Mt Norold, Southwest Tasmania, dominated here by *Isophysis tasmanica* and *Carpha curvata*, 2010. Photo: Jenny Styger

Assessment Summary

Tasmanian Alpine Sedgeland is dominated by hard-leaved monocotyledonous plants that form mats or tussocks. The vegetation type occurs on almost all the alpine habitat islands of Tasmania where it is usually associated with poorly drained ground. Much of its present area has resulted from burning of *Coniferous Heath*, so its extent is greater today than at the time of the European invasion. One plausible pathway to ecosystem collapse results in vegetation loss, while an alternative pathway involves transformation from the upward migration of lowland species in response to climate change. The status of the ecosystem type was assessed as **Near Threatened (Least Concern – Near Threatened)** based on its relatively restricted distribution, but an absence of evidence for continuing declines (Table 28).

Table 28. Summary of the Red List of Ecosystems assessment of the *Tasmanian Alpine Sedgeland*. Category ranges in parentheses reflect uncertainty in assessment under the corresponding criteria

Criteria	**A**	**B**	**C**	**D**	**E**	**Overall**
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	NT (LC-NT)	DD	DD	DD	NT (LC-NT)
Sub-criterion 2 *A, C, D: 50-year period including present & future* *B: AOO*	LC	LC	DD	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	LC	LC	DD	DD		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria

Ecosystem Description

Ecosystem Classification

In the IUCN Global Ecosystem Typology v2.0 (Keith et al. 2022a), this sub-global ecosystem type (Level 6) belongs to Ecosystem Functional Group *T6.4 Temperate alpine grasslands and shrublands* within the *Polar/alpine (Cryogenic)* biome.

Within Australia, *Tasmanian Alpine Sedgeland was* described as alpine sedgeland by Kirkpatrick (1983, 1997). Vegetation communities HSE (Highland Eastern Sedgeland), MGH (Highland Grassy Sedgeland) and HSW (Highland Western Sedgeland and Herbland) (Kitchener & Harris 2013) represent this vegetation type and are mapped.

The properties and dynamics of this vegetation type contribute to the outstanding natural significance of the Western Tasmanian Wilderness World Heritage Area so are covered by the *Environment Protection and Biodiversity Conservation Act 1999*

Distinction from similar ecosystem types

Tasmanian Alpine Sedgeland shares some of its features with *Cushion Moorland* and *Alpine Sphagnum Bog and Associated Fen*. The key distinction is that *Tasmanian Alpine Sedgeland* is dominated by hard-leaved monocotyledonous plants that form mats or tussocks, whereas *Cushion Moorland* is dominated by cushion growth forms, and *Alpine Sphagnum Bog and Associated Fen* is dominated by hummock-forming *Sphagnum* moss or wet heath shrubs and have waterlogged soils. *Tasmanian Alpine Sedgeland* is often found in a matrix of *Tasmanian Alpine Heath* and *Cushion Moorland*. *Tasmanian Alpine Sedgeland* and *Cushion Moorland* are only found in Tasmania, whereas *Alpine Sphagnum Bog and Associated Fen* is found in Tasmania and across mainland Australia

Distribution

Tasmanian Alpine Sedgeland is located between 145.47° to 148.16° longitude and between -41.18° and -43.54° latitude, occurring across more than 60 alpine habitat islands. There is a high likelihood that the vegetation type is more extensive now than prior to the British invasion, replacing areas of *Coniferous Heath* after it was burnt.

In total, *Tasmanian Alpine Sedgeland* is estimated to occupy an area of about 462.91 km^2. Spatial products used to create the present-day distribution map in this assessment (Figure 66; DPIPWE 2020) represent the most extensive and/or accurate mapping available within Tasmania for the defined bounds of *Tasmanian Alpine Sedgeland* at the time of assessment (April 2021), as advised by government representatives involved in the assessment process.

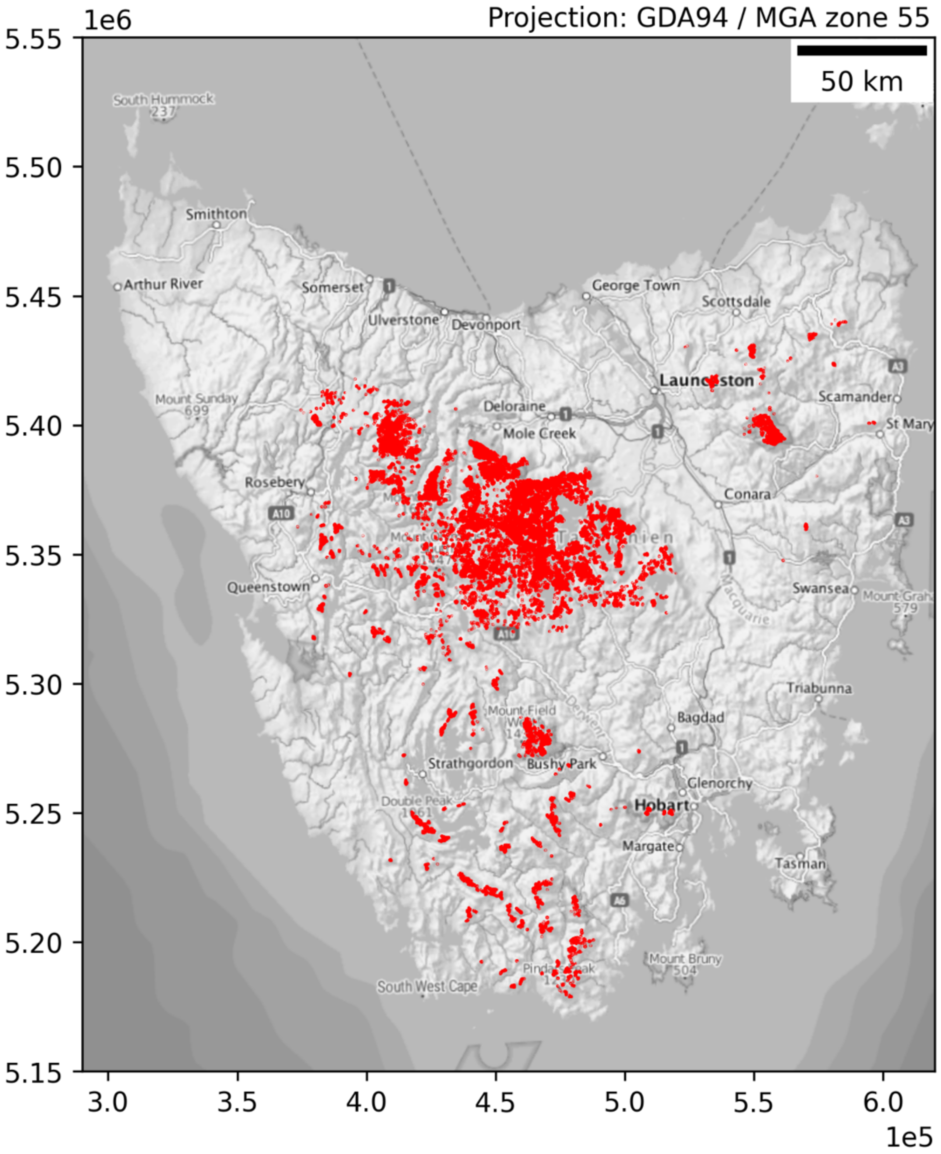


Figure 66. Distribution of the *Tasmanian Alpine Sedgeland* (red) across Tasmania.

Abiotic environment

Tasmanian Alpine Sedgeland is associated with shallow poorly drained organosols with low pH, low levels of exchangeable phosphorous and Manganese (Kirkpatrick & Bridle 1999). It occurs in areas exposed to fierce winter winds as snow cover does not persist, and is grazed by native mammals. Although able to persist in the long-term absence of fires, at least in western Tasmania, fires have expanded the cover of *Tasmanian Alpine Sedgeland* by reducing shrub cover in areas previously occupied by *Tasmanian Alpine Heath* and *Coniferous Heath* (Kirkpatrick & Bridle 1999). Although there is evidence for localised extinction of fire sensitive shrub species and vegetation transitions within alpine areas during the Holocene in response to fire (MacPhail 1986; Fletcher et al. 2014). European colonisation resulted in an increase in the extent and severity of fires in Tasmanian alpine areas (Kirkpatrick & Bridle 2013), which has favoured Tasmanian alpine sedges at the expense of shrubs (Kirkpatrick et al. 2010).

Characteristic native biota

Tasmanian Alpine Sedgeland in eastern and central mountains is dominated by varying mixtures of hard-leaved or wiry monocots including *Astelia alpina, Empodisma minus, Carpha alpina, C.*

rodwayi, Lepidosperma filiforme, Eurychorda complanata, Baloskion australe and the fern *Gleichenia alpina* (Kirkpatrick 1997; Kitchener & Harris 2013). *Isophysis tasmanica* and *Carpha curvata* are the most commonly dominant plants of the western mountain extent of *Tasmanian Alpine Sedgeland* (Kitchener & Harris 2013). Smaller forbs are scattered among the sedges and in the eastern and central mountains include *Acaena* spp., *Wahlenbergia* spp., *Senecio gunnii*, *Gunnera cordifolia*, *Gonocarpus micranthus* and *Hydrocotyle h*irta. Some areas may be grassy, including taxa such as *Poa*, *Rytidosperma* and *Microlaena*. Woody plants are usually absent from the ecosystem type, but *Richea scoparia*, *R. acerosa* and *Baeckea gunniana* may be sparsely distributed in some areas. Prostrate shrubs such as *Pentachondra pumila* and *Montitega dealbata*, or cushion plants occur in gaps. *lsophysis tasmanica* and *Carpha curvata* are the most commonly dominant plants of the western mountain extent of *Tasmanian Alpine Sedgeland*.

Tasmanian Alpine Sedgeland is grazed by native mammals, including wombats (*Vombatus ursinus*; Figure 67) and Bennet’s wallaby also known as the red- neck wallaby (*Notamacropus rufogriseus* subsp *rufogriseus*; Figure 68) and rufous-bellied pademelon *(Thylogale billardierii)*. Native rodents that utilise this habitat include the Tasmanian endemic species long-tailed mouse (*Pseudomys higginsi),* and endemic sub-species Broad-toothed rat (*Mastacomys fuscus* subsp. *fuscus*; Figure 69). Predators that include this habitat in their range include the eastern quoll (*Dasyurus viverrinus)*, the spotted-tail quoll (*Dasyurus maculatus),* Tasmanian devil (*Sarcophilus harrisii)* (NRE unpublished report). The invertebrate fauna is poorly known, a recent study comparing invertebrate fauna of burnt and unburnt subalpine grassy sedgelands demonstrates they support a diverse range of higher invertebrate taxa (Kirkpatrick et al. 1993; Driessen et al. 2023).

Figure 67. Bare nosed wombat, *Vombatus ursinus tasmaniensis,* browsing and burrowing in areas of grassy sedgeland in the Cradle Mountain - Lake St Clair National Park. Photo: Michael Driessen.

Figure 68. Bennet's wallaby (*Notamacropus rufogriseus)* in *Baloskion* dominated rushland in the Central Highlands of Tasmania. Photo: Michael Driessen.

Figure 69. Broad-toothed mouse (*Mastacomys fuscus subsp. fuscus*). Photo: Michael Driessen.

Key components, processes, and interactions

Much of the eastern extent of *Tasmanian Alpine Sedgeland* has resulted from historical removal of shrubs by fire. The western extent of *Tasmanian Alpine Sedgeland* seems less fire-dependent, although there is some evidence of expansion as a result of fires (Kirkpatrick 1997). Most of the dominant non-woody plants recover vegetatively and rapidly after fire, mainly from rhizomes. Macropods are present in the ecosystem type, but it is unclear how important they may be in maintaining the low cover of shrubs. The scarcity of shrubs, in combination with the climatic excess of precipitation over evapotranspiration, is likely promoting persistence of moisture in surface soils and the development of organosol substrates (Figure 70).

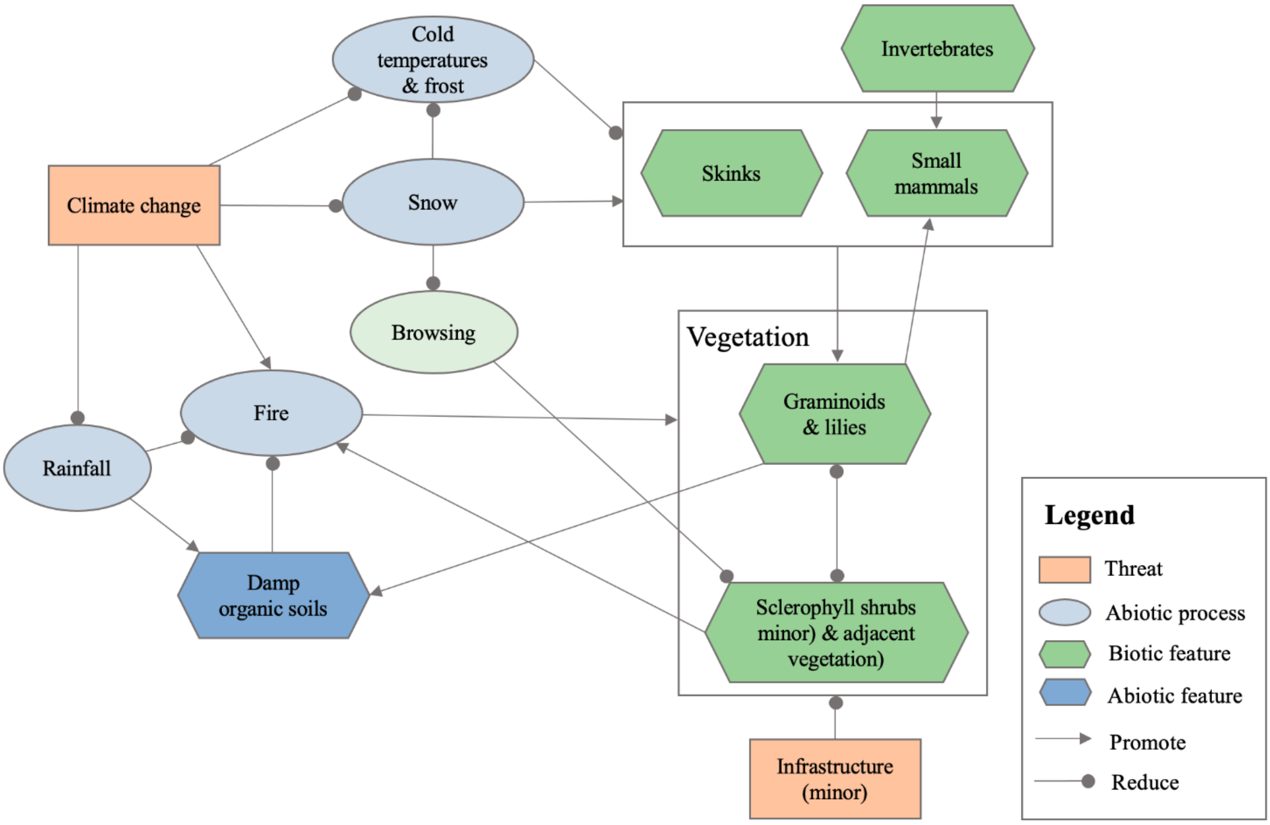


Figure 70. Conceptual diagram illustrating relationships between, and threats to, biotic and abiotic components and processes in *Tasmanian Alpine Sedgeland*.

Major threats

Fire

Burning of *Tasmanian Alpine Sedgeland* is followed by rapid vegetative recovery and may be important in maintenance of low shrub cover. But a high density of non-woody ground cover plants may limit shrub recruitment, even in the post-fire environment. Organic soils are unlikely to be incinerated in wet situations, unless shrubs are present. When substrate fires do occur, however, they will likely kill the subterranean recovery organs of the dominant non-woody plants, leading to ecosystem collapse. Recovery after substrate fires is likely to be slow, depending on dispersal of seed, seedling establishment and re-accumulation of organic substrates over at least multi-decadal time scales (Kirkpatrick 1984a; Bridle & Kirkpatrick 1997). A recent study comparing burnt and unburnt highland grassy sedgeland demonstrated the lack of recovery in vegetation cover where the organic soil horizon was combusted (Driessen et al. 2023).

Developments

Road and dam building has destroyed small areas of *Tasmanian Alpine Sedgeland*. Proposed cable cars and associated development might destroy a more, as might some ecotourism development encouraged through the State Government Expression of Interest process.

Climate warming

A warming climate might result in invasion of lowland sedges, such as *Gymnoschoenus sphaerocephalus* or *Lepidosperma filiforme*. Climatic drying might also result in shifts in species composition or oxidation of organosols (Kirkpatrick 1984a), but there is little evidence of such effects at present.

IUCN Stresses Classification

1.2 – Ecosystem degradation
1.3 – Indirect ecosystem effects

IUCN Threats Classification

1.3 – Tourism and recreation areas
4.1 – Roads & railroads
7.2 **–** Dams & Water Management/Use
8.2 – Problematic native species (encroachment)
11.1 – Habitat shifting and alteration
11.2 – Droughts
11.3 – Temperature extremes

Ecosystem collapse

Tasmanian Alpine Sedgeland collapses when the characteristic hard-leaved monocotyledonous plants are eliminated or replaced by other species or by bare ground. For example, sedge species common in lowland sedgelands (e.g., *Gymnoschoenus sphaerocephalus, Lepidosperma filiforme*) or shrubs from *Tasmanian Alpine Heath* may invade under a warming, drying climate. Alternatively, severe fire may cause the combustion of organic soils and if followed by sheet erosion could result in areas of exposed bare gravels and rock (Figure 71).

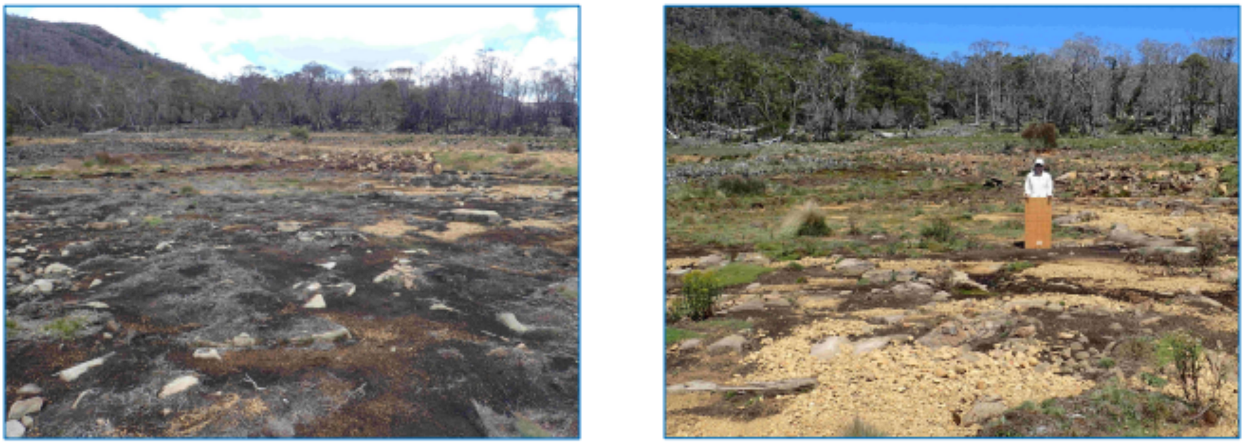


Figure 71. Area of highland grassy sedgeland one year (left) and five years (right) after a fire at Ritters Plains in 2016, which burnt patches of organic soil that subsequently resulted in localised sheet erosion, exposing gravel substrate. Photo Michael Driessen.

Tasmanian Alpine Sedgeland collapses when any of the following occur:

- *Area:* The mapped distribution declines to zero (100% loss) (Criteria A and B), or;
- *Monocot cover:* Cover of monocot plants declines to < 30% (e.g., because of post-fire erosion) (Criterion D), and/or;
- *Shrub cover:* Cover of shrubs is > 30% signalling a transition to *Tasmanian Alpine Heathland* (Criterion D).

Risk Assessment

Criterion A: Reduction in distribution

Summary

The area of this ecosystem type has increased because of the loss of large areas of *Coniferous Heath* ecosystems from fire. Sheet erosion, inundation and clearance affect trivial areas in relation to this increase. It is likely that the area of *Tasmanian Alpine Sedgeland* will increase in the next 50 years due to the future effects of fires over the next 50 years (A2), as it has in the last 50 years (A1) and since the European colonisation (A3) (Kirkpatrick & Bridle 2013). The putative expansion of the distribution of *Tasmanian Alpine Sedgeland* over the past, present, and future therefore indicates a status of **Least Concern** under sub-criteria A1, A2 and A3.

Criterion B: Restricted distribution

Summary

Tasmanian Alpine Sedgeland is assessed as **Near Threatened (Least Concern-Near Threatened)** under sub-criterion B1, and **Least Concern** under sub-criteria B2 and B3.

Methods

Under criterion B, the extent of occurrence (EOO; sub-criterion B1) and area of occupancy (AOO; sub-criterion B2) of *Tasmanian Alpine Sedgeland* were determined using existing map products from Tasmania (DPIPWE 2020; Kitchener & Harris 2013). A complete description of the data and methods used to create the current distribution map for this ecosystem type is provided in the main methods.

The number of threat-defined locations was based on encroachment by shrubs as this is the most important threat to *Tasmanian Alpine Sedgeland*

Assessment outcome

Sub-criterion B1

A minimum convex polygon encompassing *Tasmanian Alpine Sedgeland* has an area of 37,444.69 km^2. Although this estimate is within the threshold defining the Vulnerable category, Tasmanian Alpine Sedgeland meets none of the B1 sub-criteria because there is currently no continuing decline in extent of function, no threats likely to cause a continuing decline, and the ecosystem type is likely to occur at more than five threat-defined locations (see sub-criterion B3 below). Its status under sub-criterion B1 is therefore **Near Threatened (Least Concern – Near Threatened)**.

Sub-criterion B2

Tasmanian Alpine Sedgeland intersects a minimum of 205 10×10 km^2 grid cells (Figure 72). The status of the ecosystem type is therefore **Least Concern** under sub-criterion B2.

Sub-criterion B3

The most serious plausible threat to *Tasmanian Alpine Sedgeland* is encroachment by shrubs. This is likely to occur semi-independently across different mountain ranges where this ecosystem type occurs. Hence *Tasmanian Alpine Sedgeland* is likely to occur at > 5 threat-defined locations and is unlikely to collapse or become Critically Endangered within a very short period. The status of the ecosystem type is **Least Concern** under sub-criterion B3.

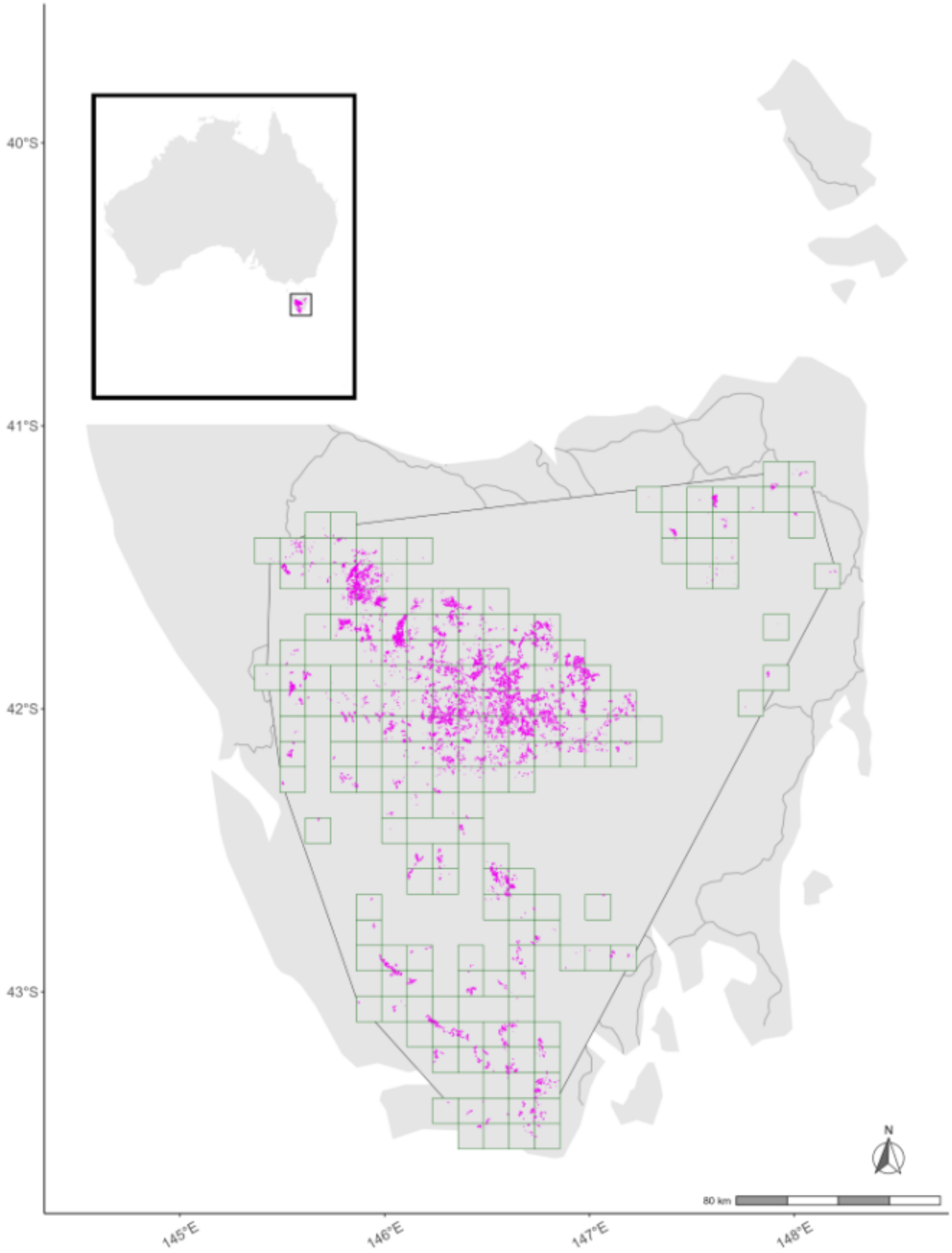


Figure 72. Map of *Tasmanian Alpine Sedgeland* (magenta polygons) showing EOO (black polygon) and AOO where the 1% rule was not applied (green squares).

Criterion C: Environmental degradation

Summary

Post-fire soil erosion is the main cause of environmental degradation in *Tasmanian Alpine Sedgeland*. This may be severe but is highly localised (Kirkpatrick et al. 2002). However, no quantitative data are available for assessment. Climate change may increase the number of growing degree days in the ecosystem type, facilitating the invasion by non-native plants from lower elevations. However, species responses to warming are poorly understood and the collapse threshold for growing degree days is highly uncertain. The status of the ecosystem type under criterion C is **Data Deficient**.

Criterion D: Disruption of biotic processes and/or interactions

Summary

Shrub encroachment and loss of monocot plant cover are the main processes with potential to disrupt biotic processes and interactions within *Tasmanian Alpine Sedgeland*. There are a few published studies on the ecosystem type (Kirkpatrick & Bridle 1999; Bridle & Kirkpatrick 2003; Kitchener & Harris 2013), however, no quantitative time series data are available for assessment. Thus, the status of the ecosystem type under criterion D is **Data Deficient**.

Identification of biotic indicator

We examined the relevance and data availability/quality for two biotic indicators to assess the risk of collapse from disruption of biotic processes and/or interactions:

- *Monocot cover*: a direct measure of the cover of the characteristic monocot species.
- *Shrub cover:* a direct measure of the transition to *Tasmanian Alpine Heath.*

Indicator: Monocot cover

Relevance to ecosystem function

Tasmanian Alpine Sedgeland is characterised by various combinations of hard-leaved or wiry monocots that form mats or tussocks (Kirkpatrick 1997). Common species include *Astelia alpina*, *Isophysis tasmanica, Empodisma minus*, *Carpha spp.*, *Lepidosperma filiforme*, *Eurycorda complanata*, *Baloskion australe*. The loss of these monocot species would lead to the loss of the ecosystem type and transition to *Tasmanian Alpine Heath.*

Selection of collapse threshold

We consider the ecosystem type to collapse when the cover of the characteristic monocot plant species declines to < 30%.

Data availability and quality

Insufficient data are available to assess this process, hence we could not adequately assess this indicator.

Indicator: Shrub cover

Relevance to ecosystem function

Shrubs can exist in *Tasmanian Alpine Sedgeland* at very sparse densities, including *Richea scoparia, R. acerosa, Baeckea gunniana, Pentachondra pumila,* and *Montitega dealbat*a (Kitchener & Harris 2013). The high density of ground-covering monocots limits shrub recruitment. Burning of shrubs and rapid post-fire regeneration by monocots may also maintain the low shrub cover (Kirkpatrick 1997). However, a warming climate may facilitate invasion by shrubs, shifting the ecosystem type to *Tasmanian Alpine Heath*.

Selection of collapse threshold

We consider the ecosystem type to collapse when the cover of shrubs increases to > 30%, indicating a transition to *Tasmanian Alpine Heath*.

Data availability and quality

Although surveys of alpine vegetation were undertaken in the 1980's (Kirkpatrick 1983, 1984a) resurveys of these areas have not yet been undertaken. Surveys for Mt Field Plateau in 1979 and 2019 (Minchin & Davies 2019; Minchin et al. 2022) are yet to be fully analysed and made available. Insufficient data are currently available to assess this process, hence we could not adequately assess this indicator.

Criterion E: Quantitative analysis of probability of collapse

Summary

No model of *Tasmanian Alpine Sedgeland* exists to estimate the probability of ecosystem collapse over the next 50 to 100 years. Its status under criterion E is therefore **Data Deficient**.

Example of *Tasmanian Alpine Sedgeland* on the Snowy Range, southwest Tasmania, 1985, dominated by *Astelia alpina*. Photo: Jayne Balmer.

Alpine-subalpine Damp Valley Grassland and Rushland

Authors

Emily Nicholson, Courtney Taylor, Ella Plumanns Pouton, Susanna Venn

Reviewers

Courtney Taylor, Matt White

Biome

T6 Polar-alpine

Functional group

T6.4 Temperate alpine grasslands and shrublands

IUCN status

Least Concern

Assessment Summary

Alpine-subalpine Damp Valley Grassland and Rushland forms a continuum of grass-dominated and rush-dominated communities, treated here as a single ecosystem type. The unifying feature of this ecosystem type is the valley bottom environment, where fluvial and aeolian processes have created gentle slopes and deep fertile soils subject to intermittent seasonal waterlogging.

Alpine-subalpine Damp Valley Grassland and Rushland faces a range of ongoing biotic and abiotic threats, including invasive species, changing fire regimes, tourism and development, and climate change. Although current threats are not anticipated to cause significant medium-term declines in this ecosystem type, many components are understudied, making it difficult to predict its response to a changing climate. *Alpine-subalpine Damp Valley Grassland and Rushland* is assessed as **Least Concern,** but many criteria are **Data Deficient** (Table 29).

Table 29. Summary of the Red List of Ecosystems assessment of the *Alpine-subalpine Damp Valley Grassland and Rushland*. Category ranges in parentheses reflect uncertainty in assessment under the corresponding criteria.

Criteria	A	B	C	D	E	Overall
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	LC	DD	DD	DD	**LC**
Sub-criterion 2 *A, C, D: 50-year period including present & future* *B: AOO*	DD	LC	DD	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	LC	LC	DD	DD		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria

Ecosystem Description

Ecosystem Classification

Alpine-subalpine Damp Valley Grassland and Rushland belongs to Ecosystem Functional Group *T6.4 Temperate alpine grasslands and shrublands* within the *Polar/alpine (cryogenic)* biome in the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022a). Under the National Vegetation Information System (NVIS) 6.0, the ecosystem type aligns with Major Vegetation Group (MVG) 21 (Other grasslands, herblands, sedgelands and rushlands) and MVG 19 (Tussock grasslands: NVIS Technical Working Group 2017). This ecosystem type is classified as multiple systems in Southeastern Australia, including Valley Grassland and Tussock Grasslands (Venn et al. 2017), Alpine Valley Grassland and Sub-alpine Valley Grassland (McDougall & Walsh 2007), Sod Tussock Grasslands (Costin et al. 2000), and *Empodisma* (Hope et al. 2012).

In Tasmania, *Alpine-subalpine Damp Valley Grassland and Rushland* comprise the wetter components of Highland Grassy Sedgeland (unit MGH) and Highland *Poa* Grassland (unit GPH; Kitchener & Harris 2013).

Distinction from similar ecosystem types

Alpine-subalpine Damp Valley Grassland and Rushland is characterised by a dominance of either tussock grasses, typically *Poa costiniana*, and/or mat-forming rushes, typically *Empodisma minus* (McDougall & Walsh 2007). This ecosystem type usually occurs within a mosaic of water-dependant ecosystem types in Australia's Alpine and sub-Alpine regions – *Alpine-subalpine Fens*, *Alpine Sphagnum Bogs and Associated Fens*, and in Tasmania, *Tasmanian Alpine Sedgeland* (a community dominated by hard leaved monocots). *Alpine-subalpine Damp Valley Grassland and Rushland* often marks the transition from permanently 'wet' vegetation and intermittently 'dry' vegetation (Hope et al. 2009), occupying the habitat niche where high soil saturation excludes many herbaceous species that occur on adjacent dry slopes, and the lack of permanent water prevents the development of bog and fen plant communities.

Distribution

Alpine-subalpine Damp Valley Grassland and Rushland occurs in the mountainous alpine and subalpine environments of south-eastern Australia (ACT, NSW, Victoria, and Tasmania). The ecosystem type is distributed between 145.48° to 148.93° longitude and between -35.46° and -43.46° latitude. In Tasmania, the distribution of *Alpine-subalpine Damp Valley Grassland and Rushland* was derived from the aerial photograph mosaic in TASVEG 4.0 (DPIPWE 2020), including grasslands on flat ground associated with lotic and lentic wetlands that occur above 800 m ASL. In New South Wales and Victoria, the lower limit of *Alpine-subalpine Damp Valley Grassland and Rushland* is approximately 1000 m, which is the approximate lower altitude limit of *Poa costiniana*.

In total, *Alpine-subalpine Damp Valley Grassland and Rushland* covers an area of approximately 511.86 km^2 (Figure 73). Spatial products used to create the present-day distribution map represent the most extensive and/or accurate mapping available within each state at the time of assessment (July 2021), as advised by representatives from each government department involved in the assessment process.

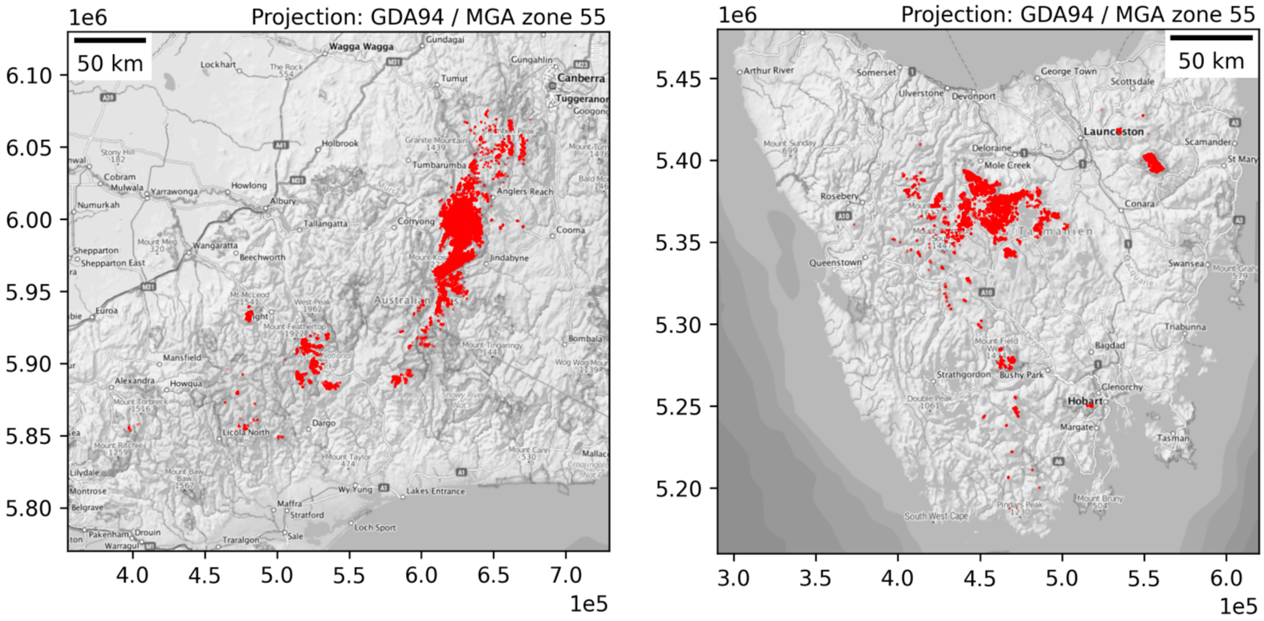


Figure 73. Distribution of the *Alpine-subalpine Damp Valley Grassland and Rushland* (red) across the Australian mainland (left) and Tasmania (right).

Abiotic environment

Alpine-subalpine Damp Valley Grassland and Rushland occurs in broad valley floors, seepage areas on gentle slopes, and on the margins of low velocity waterways subject to snow cover during winter and cold air drainage throughout the year (Costin et al. 2000; McDougall & Walsh 2007). It occupies a distinct geomorphological and edaphic habitat: alluvial flats and valley floors with fertile and typically organic soils. The soils in are typically underlain by sodden humified peats, but may accumulate peat *in lower-lying and/or colder contexts* (McDougall & Walsh 2007; Hope et al. 2012). *Alpine-subalpine Damp Valley Grassland and Rushland* occurs in areas where the soil is intermittently wet with restricted drainage. This specific hydrological niche is consistently damp, a balance between precipitation, groundwater, snowmelt, and evapotranspiration, which can be high in summer, and supports vegetation that are adapted to intermittent wetting and drying. Duration of snow cover affects growing season length, water availability and habitat for species which use the subnivean space. It is an important abiotic feature which differs between the mainland and Tasmania. On the mainland, winter snow cover may persist for weeks or months at high elevation. In Tasmania, snow cover is shorter, so the vegetation is exposed to severe winter frosts and harsh winds that limit establishment and persistence of woody species (Williams 1987).

Characteristic native biota

Flora

Grass tussocks and rush mats form the main structural components of the vegetation with – in comparison to adjacent plant communities – generally low plant diversity. The defining species for *Alpine-subalpine Damp Valley Grassland and Rushland* are *Poa costiniana* and *Empodisma minus,* although the dominant species can vary depending on geographic region and physiography. Sedge species from the genus *Carex* (which are abundant in adjacent wetland communities) may also be common. *Deschampsia cespitosa* and *Poa gunnii* are prominent grass species in frequently inundated parts of the Tasmanian distribution (Kitchener & Harris 2013). *Rytidosperma nudiflorum* sometimes co-dominates with *P. costiniana* in mainland vegetation, and *Baloskion australe* may be common where *Empodisma* dominates (Costin et al. 2000; McDougall & Walsh 2007). The dominant growth form within *Alpine-subalpine Damp Valley Grassland and Rushland* can vary between and within sites, creating spatial heterogeneity between grass-dominated (e.g., as described for Sod Tussock Grasslands in (Costin et al. 2000) and rush dominated areas (e.g., *Empodisma* moors). Weed diversity and abundance are high in some areas due to disturbance from historical cattle grazing and current presence of feral ungulates (McDougall & Walsh 2007).

Fauna

Alpine-subalpine Damp Valley Grassland and Rushland provides important food and habitat resources for a range of native mammals, reptiles, amphibians, and invertebrates (Green & Osborne 2012). Invertebrate herbivores are the most common and abundant fauna in the ecosystem type. They are likely important for ecosystem processes, but there is a paucity of research on their diversity and impacts. Phytophagous bugs are some of the most common invertebrates, followed by grasshoppers, katydids, and crickets (Green & Osborne 2012). There is also a high diversity of worms (Wilson et al. 2021). Compared with other alpine habitats, flies and wasps are relatively uncommon, likely due to the lack of floral resources.

Rodents such as the Broad-toothed rat (*Mastacomys fuscus*) and the Bush rat (*Rattus fuscipes*) are common in the sites comprised of tussock grasses and shrubs, which allow for formation of the subnivean space (Schulz et al. 2019). Skinks are common across *Alpine-subalpine Damp Valley Grassland and Rushland*, including the Southern tussock skink (*Pseudemoia pagenstecheri)* and the Alpine water skink (*Eulamprus kosciuskoi*). The Alpine she-oak skink (*Cyclodomorphus praealtus)* is largely confined to grassland areas of this ecosystem type (Green & Osborne 2012). Frogs are common in sites that contain small pools of water, including Alpine tree frog (*Litoria verreauxii alpina)* and Common eastern froglet (*Crinea signifera*). The Critically Endangered Northern Corroboree Frog (*Pseudophryne pengilleyi*) is also present*:* they lay their eggs in deep moss and grass litter. Larger grazing mammals such as wombats (*Vombatus ursinus*) and wallabies (*Notamacropus rufogriseus* in Tasmania and *Wallabia bicolor* on the mainland) maybe common in Tasmania and lower sub-alpine areas of the mainland (Bridle & Kirkpatrick 1999)In Tasmania, meso-predators such as Tasmanian devils (*Sarcophilus harrisii*) and quolls (*Dasyurus viverrinus* and *D. maculatus*) may use this ecosystem type as a hunting ground. Feral herbivores such as horses, deer, pigs, rabbits, and hares can also be common, using this ecosystem type for grazing. Feral cats and foxes may be present, particularly at lower elevations.

Key components, processes, and interactions

Hydrological regimes

Water is the primary driver influencing *Alpine-subalpine Damp Valley Grassland and Rushland*, determined by surface flows, groundwater, snow melt and topography (Figure 74). This ecosystem type relies on excess water to provide the intermittent inundation of soils. This hydrological regime creates soils that are typically too persistently wet for many mid-slope specialist herbs, grasses and shrubs and 'too dry' for the formation of wet-heath, bog and fen. *Alpine-subalpine Damp Valley Grassland and Rushland* can persist for thousands of years, as indicated by a pollen profile from Cameron Lagoon on the Central Plateau of Tasmania (Thomas & Hope 1994). This suggests that these hydrological regimes may have been relatively stable over long time periods. In areas dominated by rushes, inundation for prolonged periods can lead to the formation and accumulation of fibrous peat.

Other drivers of floral composition

Other biotic and abiotic drivers also influence floristic composition, particularly competitive exclusion by dominant species that impedes the establishment and persistence of a broader suite of species in these environments. For example, the root mats of grasses and rushes can prevent tree and shrub invasion (Fensham & Kirkpatrick 1992). Cold air accumulation in the valley bottoms (where this ecosystem type typically occurs) can cause severe frosts, limiting frost-intolerant woody species. Periodic stressors such as drought and insect attack can cause large areas of *Poa* dieback (Costin et al. 2000). Areas of dieback are initially colonised by forbs, succeeding to a dominance of grasses, cyclically operating over multi-year cycles (Costin et al. 2000).

Vegetation shifts

Very few studies have examined the internal vegetation dynamics within *Alpine-subalpine Damp Valley Grassland and Rushland*. In one exception, Kirkpatrick et al. (2019) found that rushland was expanding into grassland. Such expansion is slow and may represent the growth of individual rush plants (Kirkpatrick et al. 2019). Limited differences in environmental factors and soil properties between the grassland and rushland patches suggests that these changes may be cyclical; however, limited long-term data exists and these increases may be due to a release from fire and pastoral grazing (Kirkpatrick et al. 2019). Importantly, the ecosystem type may expand into *Alpine Sphagnum Bog and Associated Fen* that have been degraded at the margins by fire and grazing (Hope et al. 2012; Hope & Nanson 2015); this may become more likely under a warmer, drier climate. *Empodisma minus* has been observed colonising former *Sphagnum* hummocks post-fire and *Poa* species can colonise dried out peat (Whinam & Chilcott 2002). Therefore, this ecosystem type is one of the degradation states of *Alpine-subalpine Bog and Associated Fens* and *Alpine-subalpine Fen*.

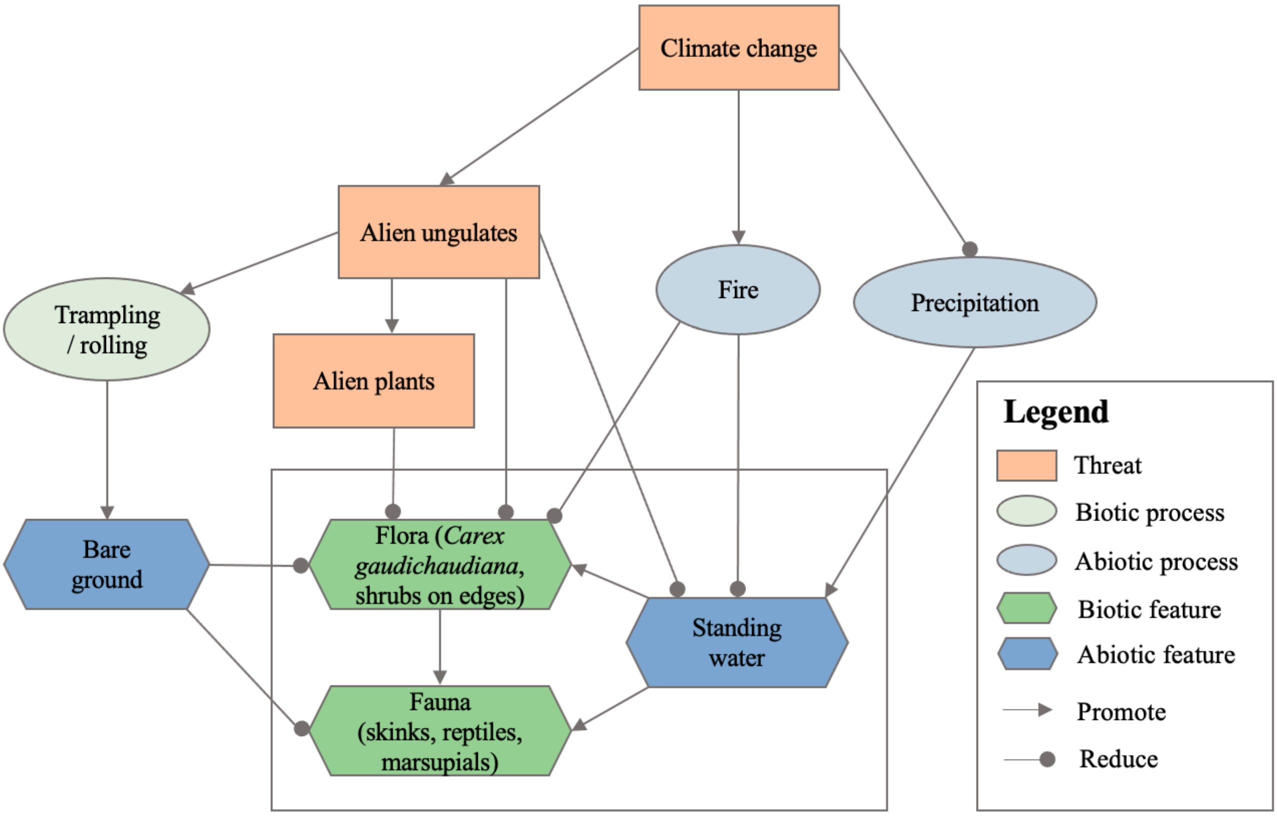


Figure 74. Conceptual diagram illustrating relationships between, and threats to, biotic and abiotic components and processes in *Alpine-subalpine Damp Valley Grassland and Rushland*. Key components are highlighted in the rectangle.

Major threats

Alien ungulates and plants

Feral ungulate grazing and trampling can simplify this ecosystem type, replacing tussock grasses and rushes with short turf and exotic plants, particularly pasture grasses (Lembrechts et al. 2016; Tolsma & Shannon 2018). This negatively impacts dependent fauna, such as the Broad-toothed rat and Alpine water skink, which require a dense and complex vegetation structure (Eldridge et al. 2019; Driscoll et al. 2019). The feral horse (*Equus calballus*) population is currently estimated at over 25,000 individuals in the Australians Alps region (Cairns 2020). Horse damage is particularly problematic in Kosciuszko National Park (New South Wales) and the Nunniong Plateau (Victoria), where populations are concentrated. Trampling by horses has been shown to disrupt peat stores, shifting them from carbon sinks into carbon sources and disrupting soil properties (Treby & Grover 2023). Trampling by feral ungulates in wetlands and riparian areas, primarily sambar deer (*Rusa unicolor*) in mainland areas and fallow deer (*Dama dama*) in Tasmania (Tolsma & Shannon 2018), can also alter hydrological systems. Further, excavation and wallowing by pigs can create patches of bare soil, causing desiccation and providing opportunities for weed invasion (McDougall & Walsh 2007). Invasion by alien plants, in particular competitive pasture grasses and herbs, can change the composition and structure of the vegetation and reduce the habitat suitability for native fauna (McDougall et al. 2005; Eldridge et al. 2019).

Climate Change

The trajectory of *Alpine-subalpine Damp Valley Grassland and Rushland* under a changing climate is not clear. But climate change poses a long-term threat, as warmer and drier climates (including more frequent droughts) are likely to alter the hydrology of the ecosystem type (Worboys et al. 2011). Reduced snow-cover will impact species that rely on the subnivean space for insulation from freezing winter temperatures. The predicted higher temperatures and lower summer precipitation may lead to an expansion of *Alpine-subalpine Damp Valley Grassland and Rushland* into typically wetter ecosystem types (e.g., *Alpine and subalpine Fen*) but simultaneously lead to contraction at the up-slope edges of the system, where drying may lead to invasion from shrubs and dryland grasses (Worboys et al. 2011; Wahren et al. 2013). Drying, in combination with increased fire frequency, could cause a transition to a species poor *Alpine-subalpine Open Grassy Heathland and Herbfield,* or a novel ecosystem type that is a drier, more degraded version of its current state.

Alpine-subalpine Damp Valley Grassland and Rushland is resilient to low severity fire that does not disrupt the underlying peat soil structure. The cover of dominant species can recover within ~12 months post fire (Walsh & Mcdougall 2004; McDougall & Walsh 2007; Hope et al. 2009; McDougall et al. 2023). However, increasing drought conditions combined with increased fire frequency and severity represent a substantial threat, as vegetation cannot recover from overly frequent fire in tandem with drought (Hope & Nanson 2015).

IUCN Stresses Classification

1.2 – Ecosystem degradation
1.3 – Indirect ecosystem effects
2.3 – Indirect Species Effects

IUCN Threats Classification

2.3.2 – Small-holder grazing, ranching or farming
6.1 – Recreational Activities
7.1.1 – Increase in fire frequency / intensity
8.1.2 – Invasive non-native species
11.1 – Habitat shifting and alteration

11.2 – Droughts
11.3 – Temperature Extremes

Ecosystem collapse

The collapsed states of *Alpine-subalpine Damp Valley Grassland and Rushland* is not easy to predict, as it is unlikely to degrade to a significantly different vegetation type. The collapsed state is likely to be a drier, weedy state with increased bare ground, dominated by novel grass species.

Alpine-subalpine Damp Valley Grassland and Rushland collapses when any of the following occur:

1) *Area:* The mapped extent declines to zero (100 % loss) (Criteria A and B), and/or;
2) *Bare ground*: Cover of bare ground is >30 %, thereby providing inadequate protection against soil loss (Criterion C), and/or;
3) *Vegetation cover:* the dominant species (*Empodisma minus* combined with *Poa costiniana* on the mainland, or *Poa gunni* and *Deschampia caespitosa* in Tasmania) make up < 50% of vegetation cover, relative to other groups (e.g., shrubs, forbs or cushion) (Criterion D).

Risk Assessment

Criterion A: Reduction in distribution

Summary

Changes in the distribution of *Alpine-subalpine Damp Valley Grassland and Rushland* in the last 50 years are poorly documented. *Alpine-subalpine Damp Valley Grassland and Rushland* may have increased in distribution as bogs degrade into this ecosystem type after damage caused by fire or grazing (both current feral species and historical cattle grazing; Hope & Nanson 2015). Conversely, the extent of *Alpine-subalpine Damp Valley Grassland and Rushland* may have declined due to the co-occurrence of high severity fires and drought, or infrastructure development, although the overall impact on the extent is likely to be negligible. Prior to this assessment, *Alpine-subalpine Damp Valley Grassland and Rushland* was not well mapped, partly due to occurrence within a mosaic of water dependent communities, which are frequently grouped together in mapping exercises. However, it is unlikely that the ecosystem type has decreased in extent by more than 30% in the last 50 years or since European invasion, therefore is listed as **Least Concern** under sub-criterion A1 and A3. In the future, increases in distribution may continue at small scales as climate change leads to more *Alpine Sphagnum Bog and Associated Fen* transitioning to this ecosystem type. However, reduced water inputs and associated invasion by dryland plant species may lead to a reduction in distribution. Overall, the extent to which the distribution may change in the future is unclear and thus the assessment is **Data Deficient** under sub-criterion A2.

Criterion B: Restricted distribution

Summary

Alpine-subalpine Damp Valley Grassland and Rushland is assessed at **Least Concern** under all sub-criterion of criterion B.

Methods

Under criterion B, the extent of occurrence (EOO; B1) and area of occupancy (AOO; B2) of *Alpine-subalpine Damp Valley Grassland and Rushland* were determined using a combination of existing map products from Victoria (DELWP 2021) and New South Wales (Gellie 2005; Hope et al. 2012). These data were augmented by expert interpretation of aerial photography in December 2019 in Tasmania. A complete description of the data and methods used to create the current distribution map for this ecosystem type is provided in the main methods.

The number of threat-defined locations (B3) was based on drought and fire, as these processes combined are the most important stochastic threat.

Assessment outcome

Sub-criterion B1

The Extent of Occurrence (EOO) of *Alpine-subalpine Damp Valley Grassland and Rushland* is estimated at 157,076.95 km^2 (Figure 75). The risk status is **Least Concern** under sub-criterion B1.

Sub-criterion B2

Alpine-subalpine Damp Valley Grassland and Rushland occupies 188 10×10 km grid cells (Area of Occupancy, AOO; Figure 75). The risk status is **Least Concern** under sub-criterion B2.

Sub-criterion B3

The most plausible threat to *Alpine-subalpine Damp Valley Grassland and Rushland* distribution is prolonged periods of drought in combination with more frequent fires (return intervals of 3-5 years based on dominant species recovering from low severity fires). However, because drought and fire are likely to occur unevenly across the distribution of this landscape and thus not affect all the extent at sufficiently short intervals to cause collapse, the status of the ecosystem type under sub-criterion B3 is **Least Concern.**

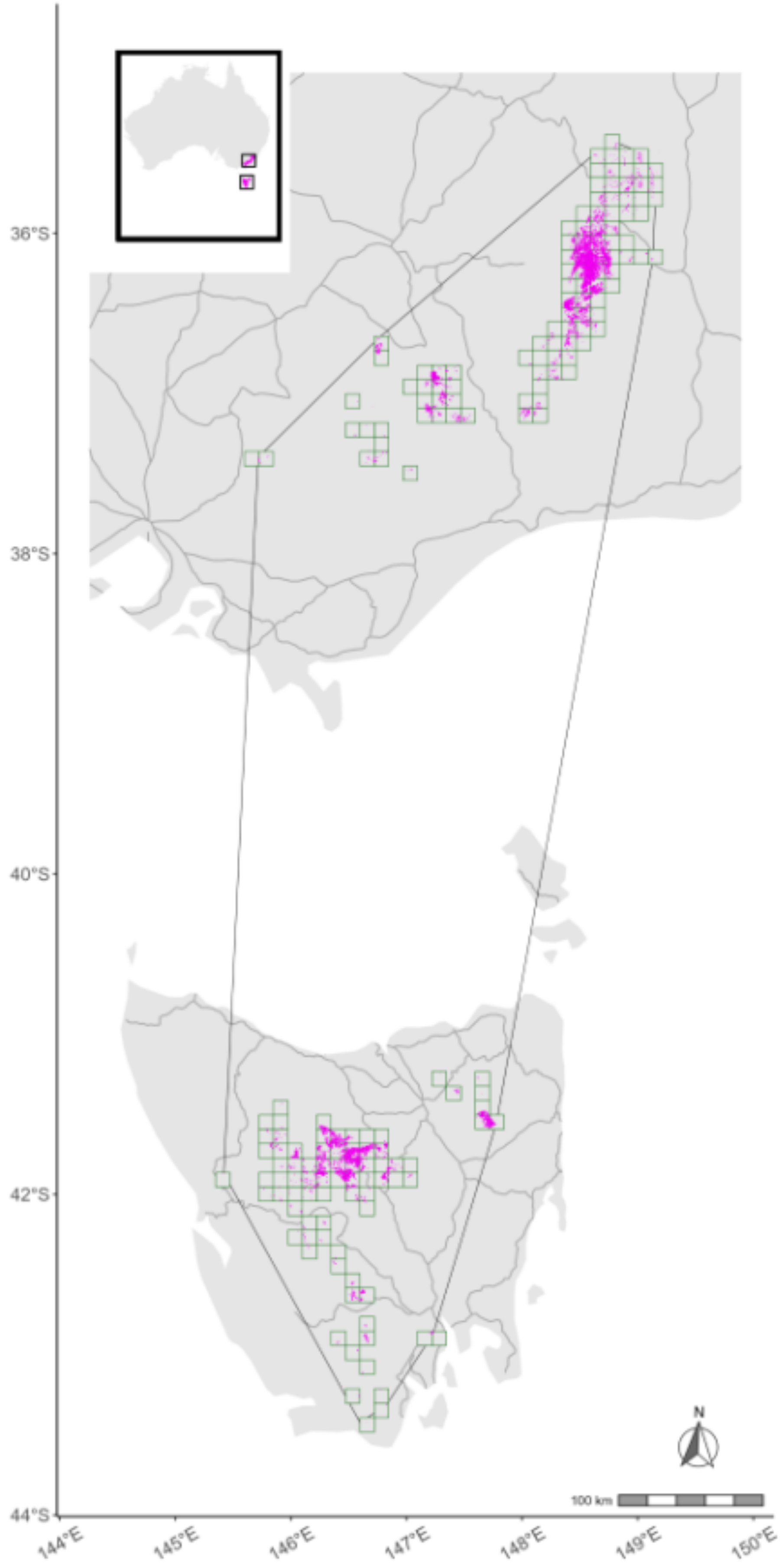


Figure 75. Map of *Alpine-subalpine Damp Valley Grassland and Rushland* (magenta polygons) showing EOO (black polygon) and AOO where the 1% rule was not applied (green squares).

Criterion C: Environmental degradation

Summary

Alpine-subalpine Damp Valley Grassland and Rushland is susceptible to environmental degradation related to climate change and impacts from feral ungulates. There is scarce data to base the assessment of current (sub-criterion C1), future (sub-criterion C2) and historical (sub-criterion C3) environmental degradation in this ecosystem type. Based on available knowledge, the indicators are assessed as **Data Deficient** under all sub-criteria.

Identification of abiotic indicator

We examined the data availability for three abiotic indicators to assess the risk of collapse from environmental degradation.

- *Bare ground cover*: a measure of groundcover lacking vegetation, but including leaf litter, rocks, and bare soil.
- *Soil moisture*: a measure of the moisture balance needed to support the characteristic vegetation.
- *Number of dry months*: a measure of this ecosystem type's vulnerability to collapse following fire.

Indicator: Bare ground

Relevance to ecosystem function

Fire and trampling by feral ungulates can increase bare ground and reduce vegetation cover. If vegetation does not recover, bare ground can lead to erosion and soil loss, and may disrupt hydrology, carbon storage, and cause weed invasion.

Selection of collapse threshold

Cover of bare ground is typically very low (< 5%) in Australian Alpine grassy ecosystems (Williams et al. 2014). Based on this, *Alpine-subalpine Damp Valley Grassland and Rushland* would be considered collapsed if cover of bare ground was > 30%.

Data availability and quality

While there are no temporal data to quantify changes in bare ground in this ecosystem type, it is unlikely to have occurred in ≥ 30 % of its extent in the last 50 years or since European invasion and there is no indication that it will increase. However, this ecosystem type is **Data Deficient** for this indicator.

Indicator: Soil moisture

Relevance to ecosystem function

Alpine-subalpine Damp Valley Grassland and Rushland is characterised by intermittent periods of wetting and drying. Waterlogged soils support the dominant species *Poa* spp. and *Empodisma minus* and allow for peat accumulation. Soil moisture would likely be a good indicator for overall moisture balance in this system. Climate change is likely to alter the moisture balance, disrupting the hydrological niche that characterises this system.

Data availability and quality

No data currently exist to determine the range of appropriate soil moisture values for this system. Therefore, this ecosystem type is **Data Deficient** for this indicator.

Indicator: Number of dry months

Relevance to ecosystem function

Alpine-subalpine Damp Valley Grassland and Rushland can recover quickly from low intensity fire events as the dominant vegetation can resprout. However, prolonged periods of drought combined with fire events can cause the underlying peat and root mass structure to be destroyed, preventing resprouting and disrupting water holding capacity. Number of dry months could act as a proxy measure of this ecosystem type's vulnerability to collapse following fire.

Data availability and quality

No data currently exist to quantify the threshold of dry months that would result in peat fires in this system. Therefore, this ecosystem type is **Data Deficient** for this indicator.

Criterion D: Disruption of biotic processes and/or interactions

Summary

Directional change in plant species composition (identity and abundance) is the most relevant indicator of biotic disruption in *Alpine-subalpine Damp Valley Grassland and Rushland.* The available data are insufficient to draw a conclusion about rates of biotic disruption. Hence the status of the ecosystem type under criterion D is **Data Deficient.**

Identification of biotic indicator

We examined the relevance and data availability for one biotic indicator to assess the risk of collapse from disruption of biotic processes and/or interactions.

- *Plant species composition:* a measure of the relative cover grasses and rushes.

Indicator: Plant species composition

Relevance to ecosystem function

Alpine-subalpine Damp Valley Grassland and Rushland is characterised by high cover of the dominant monocot species *Poa costiniana* and *Empodisma minus (and Deschampsia cespitosa* in Tasmania) which are well adapted to the intermittently water-logged and peaty soils. The ratio of grass to rush cover can differ across the distribution of this ecosystem type. Climate change may lead to changes in the water balance of this ecosystem type and hence reduce the dominant species.

Data availability and quality

There are no time series data on vegetation composition change to support assessment of this indicator.

Criterion E: Quantitative analysis of probability of collapse

Summary

No stochastic models of the ecosystem type are available and there are presently insufficient data to quantitatively estimate the probability of collapse of *Alpine-subalpine Damp Valley Grassland and Rushland* within the next 50 to 100 years. Therefore, the risk status is **Data Deficient** under criterion E.

Photo: (Owen Salkin 2007).

Terrestrial-Freshwater transitional biome

Alpine Sphagnum Bog and Associated Fen near Lake Ina, Central Plateau, Tasmania. Photo: Grant Dixon.

Alpine *Sphagnum* Bog and Associated Fen

Authors

Arn Tolsma, David Keith, Jose Ferrer-Paris, Emily Nicholson, Jessica Rowland, Jane Balmer, Keith MacDougall, Tracey Regan, Ella Plumanns Pouton, Chloe Sato, Jennie Whinam.

Reviewers

Arn Tolsma, Courtney Taylor, Anikó B. Tóth

Biome

TF1 Palustrine wetlands

Functional group

TF1.5 Boreal, temperate and montane peat bogs

IUCN status

Mainland Australia: Endangered (plausible range Endangered – Critically Endangered)
Tasmania: Endangered (plausible range Endangered – Endangered)
Australia-wise: Vulnerable (plausible range Least Concern to Vulnerable)

Alpine Sphagnum Bog and Associated Fen on the Bogong High Plains. Photo: Joslin Moore.

Assessment Summary

Alpine Sphagnum Bog and Associated Fen form a groundwater-dependent and hydrologically connected mosaic. The ecosystem type spans a total mapped area of 170 km^2 across two disjunct regions: the mainland alps where it occurs at elevations typically above 1000 – 1400 m; and in Tasmania, where it occurs above 800 m elevation. The biotic and abiotic characteristics of the bogs differ to varying degrees between these two regions. Therefore, we assessed risks to these two ecosystem types, as well as risks to the combined type for the entire Australian distribution, consistent with the circumscription of the threatened ecological community, *Alpine Sphagnum Bog and Associated Fen*, listed under Australian legislation. Across its range in Australia, *Alpine Sphagnum Bog and Associated Fen* is defined by hummock-forming *Sphagnum* moss (typically > 30% cover) alongside wet heath shrubs, sedges, and cordrushes, but can vary structurally from *Sphagnum*-dominated to shrub-dominated. The most serious threats to *Alpine Sphagnum Bog and Associated Fen* are increased frequency of fire, climate change, and introduced ungulates (especially feral horses), but on the mainland they exhibit major legacies of overgrazing by domestic livestock.

The status of *Alpine Sphagnum Bog and Associated Fen* varies between the three assessment units (Table 30). On mainland Australia, alpine bogs are **Vulnerable** (plausible range Vulnerable – Endangered) due to their restricted distribution and evidence of ongoing declines and threatening processes (criterion B1ai+ii; c). In Tasmania, alpine bogs are **Endangered** (plausible range Endangered – Endangered) due to their restricted distribution and evidence of ongoing declines and threatening processes (criterion B1ai+ii; c). The status of all alpine bogs *in Australia* is **Least Concern** (plausible range Least Concern to Vulnerable). The averaging effects and different underlying causes of risk on the mainland and Tasmania result in a lower threat status for the combined ecosystem type.

Table 30. Summary of the Red List of Ecosystems assessment of *Alpine Sphagnum Bog and Associated Fen* (a) on mainland Australia, (b) in Tasmania, and (c) in Australia. Category ranges in parentheses reflect uncertainty in assessment under the corresponding criteria.

Criteria: **a) Mainland Australia**	A	B	C	D	E	Overall
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	VU (VU-VU)	NT (LC-EN)	LC (LC-VU)	DD	**VU** **(VU-EN)**
Sub-criterion 2a&b *A, C, D: 50-year period including present & future* *B: AOO*	DD	LC (LC-LC)	LC (LC-VU)	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	LC (LC-VU)	LC (LC-VU)	LC (LC-LC)	DD		
Criteria: **b) Tasmania**	A	B	C	D	E	Overall
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	EN (EN-EN)	LC (LC-VU)	DD	DD	**EN** **(EN-EN)**
Sub-criterion 2a&b *A, C, D: 50-year period including present & future* *B: AOO*	DD	VU (VU-VU)	LC (LC-NT)	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	LC (LC-LC)	VU (LC-VU)	LC (LC-VU)	DD		
Criteria: **c) Australia**	A	B	C	D	E	Overall
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	LC	LC	LC (LC-VU)	DD	DD	**LC** **(LC-VU)**
Sub-criterion 2a&b *A, C, D: 50-year period including present & future* *B: AOO*	DD	LC	LC (LC-NT)	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	LC (LC-NT)	LC	LC (LC-LC)	DD		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria

Ecosystem Description

Ecosystem Classification

Alpine Sphagnum bogs were assessed at two different scales. Compositional differences in biota between mainland and Tasmanian bogs reflect biogeographic isolation since the Tertiary origin of mountains on the mainland, albeit with some dispersal resulting in a portion of shared biota between the two regions. Bogs in these two regions also differ in terms of climatic drivers, with those in Tasmania at latitudes exposed to stronger oceanic influence and stronger influence of moisture-bearing westerly winds, with resulting effects on fire regimes. Therefore, mainland and Tasmanian bogs were described and assessed as separate ecosystem types. In addition, a combined national ecosystem type encompassing mainland and Tasmanian alpine bogs is currently listed as the "*Alpine Sphagnum Bogs and Associated Fens*" ecological community under the Commonwealth *Environmental Protection and Biodiversity Conservation Act 1999*. The Endangered status of this combined ecological community was based on criteria (Threatened Species Scientific Committee 2009) that differ from the current international standard (IUCN 2024). Therefore, we undertook an additional Red List assessment of the combined ecological community to compare and update the status of the current statutory listing. In this section, we provide a general description of the combined *Alpine Sphagnum Bogs and Associated Fens* for Australia and identify biotic and abiotic properties that distinguish the two ecosystem types on mainland Australia and Tasmania, respectively. The three assessment units are labelled as follows:

- *Alpine Sphagnum Bogs and Associated Fens on mainland Australia;*
- *Alpine Sphagnum Bogs and Associated Fens in Tasmania;* and
- *Alpine Sphagnum Bogs and Associated Fens in Australia.*

Informally, we refer to these has mainland, Tasmanian and Australian bogs, respectively.

In the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022a), *Alpine Sphagnum Bogs and Associated Fens* is a sub-global ecosystem type (Level 6) belonging to Ecosystem Functional Group *TF1.6 Boreal, temperate, and montane peat bogs* within the *Palustrine Wetlands* biome.

In Victoria, *Alpine Sphagnum Bog and Associated Fen* encompasses two threatened ecological communities listed under the *Flora and Fauna Guarantee Act 1988* (FFG); "Alpine Bog Community" and "Fen (Bog Pool) Community" (DELWP 2018a); including all or parts of the following mapped Ecological Vegetation Communities (EVCs) (Threatened Species Scientific Committee 2009): 44 (Sub-alpine Treeless Vegetation), 171 (Alpine Fen), 210 (Sub-alpine Wet Heathland), 211 (Sub-alpine wet heathland/alpine fen mosaic), 288 (Alpine Valley Peatland), 917 (Sub-alpine wet sedgeland) and 1011 (Alpine peaty heathland).

In NSW, the ecosystem is listed under the *Biodiversity Conservation Act 2016* as part of a broader community called "Montane peatlands and swamps of the New England Tableland, NSW North Coast, Sydney Basin, South East Corner, South Eastern Highlands and Australian Alps Bioregions" (TSSC 2018). The ecosystem type assessed here corresponds with the "Alpine Bogs and Fens" vegetation class (Keith 2004) and higher-altitude examples of "Montane Bogs and Fens" in the Australian Alps bioregion (see *Distribution* section).

In the ACT, *Alpine Sphagnum Bog and Associated Fen* are listed under the *Nature Conservation Act 2014* as "High Country Bogs and Associated Fens" (EPSD 2019). In Tasmania, the ecosystem type is listed under the Nature Conservation Act 2002 as Community 36, "Sphagnum Peatland" (DPIPWE 2018), and includes "Sphagnum Peatland" grading to "Alpine Sedgeland" (Tasmania) (Harris & Kitchener 2005).

Distinction from similar ecosystem types

Small pools and fens are often integral components of *Alpine Sphagnum Bog and Associated Fen* and are thus included in the definition of the ecosystem type. Larger, stand-alone fens were assessed as a separate ecosystem type, *Alpine-subalpine Fen*. That ecosystem type is differentiated from *Alpine Sphagnum Bog and Associated Fen* by permanent or periodic standing water in large pools (cf. waterlogged soils), dominance of *Carex* and similar non-woody graminoid species, an absence of *Sphagnum,* while woody plants are absent or account for relatively sparse cover (Venn et al. 2017). *Alpine Sphagnum Bog and Associated Fen* may also occur in mosaics with two other related ecosystem types: *Alpine-subalpine Damp Valley Grassland and Rushland* (on mainland Australia and Tasmania), a continuum of grass-dominated and rush-dominated communities found in deep fertile soils subject to intermittent seasonal waterlogging in valley bottoms; and *Tasmanian Alpine Sedgeland* (Tasmania only), an ecosystem type defined by hard-leaved monocots that form mats or tussocks. On both the mainland and in Tasmania, *Alpine Sphagnum Bog and Associated Fen* transitions along altitudinal gradients to montane and lowland bogs, which lack alpine endemic biota and generally have a much-reduced Sphagnum component (Jarman & Kantvilas 1988; Keith 2004). The lower elevation bogs also include many species characteristic of milder temperate climates, and are generally more fire prone.

Distribution

Alpine Sphagnum Bog and Associated Fen is found in alpine, subalpine and high-montane regions of NSW, ACT, Victoria, and Tasmania (DOTE 2015) (Figure 76) where there is regular seasonal snow cover. On mainland Australia, it occurs primarily within the Australian Alps bioregion, and in Tasmania, it occurs in the Tasmanian Central Highlands, Tasmanian Southern Ranges and Ben Lomond bioregions (SEWPaC 2012), between 145.80° and 148.98°E longitude and between -35.48° and -42.89°S latitude. *Alpine Sphagnum Bog and Associated Fen* is currently mapped in an area of 172.87 –173.78 km^2 in Australia (Figure 76), including 80.06 – 80.10 km^2 in NSW, 6.50 km^2 in the ACT, 44.37 – 44.44 km^2 in Victoria (combined area of 130.93 – 131.04 km^2 on the mainland), and 41.94 – 42. 74 km^2 in Tasmania based on maps available from state government agencies (see Methods in main text). *Alpine Sphagnum Bog and Associated Fen* occurs naturally as patches within a matrix of other ecosystem types. Patch sizes vary from less than 0.1 ha to more than 100 ha (Tolsma 2009).

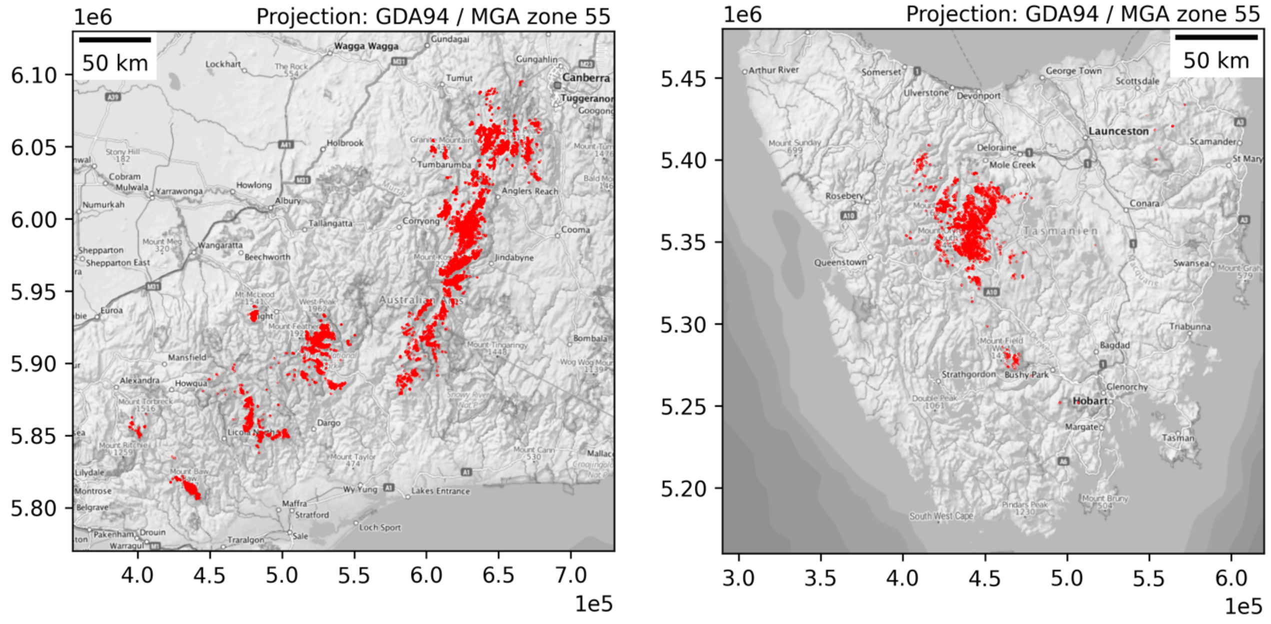


Figure 76. Distribution of the *Alpine Sphagnum Bog and Associated Fen* (red polygons) across the Australian mainland (left) and Tasmania (right). Basemap from TopPlusOpen/Grey (© GeoBasis-DE / BKG 2025).

Abiotic environment

Alpine Sphagnum Bog and Associated Fen occurs in climates where there is a surplus of precipitation over evapotranspiration for much of the year and where there is seasonal snow cover. These conditions may occur at elevations above 800 m above sea level (asl) in Tasmania and 1000 m asl on the mainland, a lower altitude than many other alpine ecosystem types, due to cold air drainage and microclimate effects within bogs. However, in a transitional zone between 1000 m and 1400 m asl on the mainland, a mixture of alpine and montane biota inhabits the bogs. Alpine bogs vary in topographic position, slope and water availability (White 2009), being found in permanently wet areas, such as along streams, drainage lines, valley edges and valley floors, and at seepage lines on slopes (Costin et al. 2000) (McDougall & Walsh 2007) (Lawrence et al. 2009). Australian bogs are characteristically acidic, nutrient-poor, but minerotrophic (fed by water filtered through mineral soils), in contrast to acidic, nutrient-poor ombrotrophic bogs (fed by precipitation) common in the northern hemisphere (Wimbush 1970; Rydin & Jeglum 2013). A small portion of Australian bogs are ombrotrophic due to mounded accumulation of peat and *Sphagnum*. In all bogs, the water table is at or near the surface for most of the year, due to abundant groundwater and impeded drainage (Whinam et al. 2003), which limits soil oxygen and microbial activity and promotes accumulation of partly-decomposed organic matter that may form peat (Hope et al. 2009, 2012). The term "peatland" usually refers to terrestrial sediments, where the depth of peat is greater than 30 cm (Hope et al. 2012). However, peat depth in many Australian *Alpine Sphagnum Bog and Associated Fen* is less than 30 cm.

The absorptive properties of *Sphagnum*, some graminoids and peat regulate the retention and flow of ground and surface water, conferring important functional roles of bogs in modulation of stream flows (Ashton & Williams 1989), filtering nutrients, sediments and pathogens, adding organic carbon, and buffering capacity that are important for downstream ecological processes (Silvester 2006, 2007, 2009). Peat development can vary substantially up to several meters deep, making bogs a valuable sink for carbon storage (Hope et al. 2012). *Sphagnum* and some graminoids in bogs act as ecosystem engineers by maintaining a very acidic pH level (3.5-4.5), which limits the organisms that can survive in *Alpine Sphagnum Bog and Associated Fen* (van Breemen 1995; Hope et al. 2012).

Characteristic native biota

Sphagnum cristatum forms hummocks throughout mainland and Tasmanian bogs, while *Sphagnum australe* and *S. falcatulatum* may be common in Tasmania (Figure 77). Other characteristic plant species that occur in both mainland and Tasmanian bogs include: Cord rushes *Empodisma minus* and *Baloskion australe,* the sedge *Carex gaudichaudiana,* and tussock grasses *Poa costiniana* on the mainland (Hope et al. 2012) and *P. labillardieri* in Tasmania (Whinam et al. 2001). Forbs such as *Asperula gunnii* and *Astelia alpina,* and the fern *Blechnum penna-marina*, occur in both mainland and Tasmanian alpine bogs, but other forbs are endemic to the mainland including *Astelia psychrocharis, Celmisia tomentosa, Dichosciadium ranunculaceum* var. *ranunculaceum*, *Gingidia algens* and *Euphrasia eichleri* (Costin et al. 2000), while the forbs such as *Celmisia asteliifolia, Pappochroma stellatum* and *Milligania densiflora,* and the fern *Gleichenia alpina* are endemic to Tasmania (Whinam et al. 2001). *Baeckea gunnii, Leptospermum lanigerum* and *Oxylobium ellipticum,* are widespread shrubs in both mainland and Tasmanian alpine bogs, but a number of woody species are either endemic to the mainland or largely so (*Epacris glacialis, Epacris paludosa, E. celata, Richea continentis*) or endemic to Tasmania (*Richea gunnii, R. scoparia*, *R.*

acerosa, *Epacris serpyllifolia*, and in restricted areas, *Athrotaxis cupressoides*) (Whinam et al. 2001). The shrub species include those that regenerate after fires from either seed or by resprouting (Walsh & Mcdougall 2004). These shrubs have varying recovery times after fire, and this combined with hydrology, helps drive the dynamics of the ecosystem type (Clarke et al. 2015).

Figure 77: Clockwise from top left are *Alpine Sphagnum Bog and Associated Fen*: dominated by *Sphagnum* and *Gleichenia alpina*, near Lake Ina, Central Plateau, Tasmania (Photo: Grant Dixon). Shrub-dominated with *Epacris paludosa* (white flowers), Snowy Range, Victoria (Photo: Arn Tolsma); in Walls of Jerusalem National Park, Tasmania, flanked by *Athrotaxis cupressoides* (Photo: Jennie Whinam); Shrub-dominated form with *Richea scoparia*, Walls of Jerusalem National Park, Tasmania (Photo: Grant Dixon).

Alpine Sphagnum Bog and Associated Fen provides significant habitat for several endemic and threatened fauna species. In mainland bogs, these include the nationally listed Critically Endangered Southern corroboree frog (*Pseudophryne corroboree)* and Northern corroboree frog (*Pseudophryne pengilleyi*), the Endangered Baw Baw Frog (*Philoria frosti*), which use *Alpine Sphagnum Bog and Associated Fen* for both breeding and hibernation, and the Vulnerable Alpine tree frog (*Litoria verreauxii alpina*), which uses the ecosystem type for breeding. Other common frog species also use the ecosystem type for breeding, such as the Common eastern froglet (*Crinia signifera*). In mainland bogs, several skink species are founding in *Alpine Sphagnum Bog and Associated Fen,* including the nationally Endangered Alpine she-oak skink (*Cyclodomorphus praealtus*), the Alpine water skink (*Eulamprus kosciuskoi*; Steane et al. 2005) and the Alpine bog skink (*Pseudomoia cryodroma*), which are Critically Endangered and Endangered respectively under Victoria's Advisory List of Threatened Vertebrate fauna (DSE 2013). The ecosystem type provides habitat for spiny crayfish *Euastacus crassus* and *E. rieki*, and several native galaxiid fish species, including the nationally Endangered Barred galaxias (*Galaxias fuscus*).

Key components, processes, and interactions

Hydrological regimes

Both on the mainland and in Tasmania, water is the major driver of bog formation, condition, and persistence (Figure 78). The overall water balance is regulated by precipitation, groundwater availability, and evapotranspiration. Australian *Alpine Sphagnum Bog and Associated Fen* occur in elevated regions receiving high orographic precipitation and comparatively low evapotranspiration due to cloud cover and low temperatures. They are minerotrophic, and thus only occur where there is a reliable supply of groundwater (Wimbush 1970; Wahren et al. 1999; Whinam et al. 2003). Groundwater availability is determined by precipitation and recharge, and capacity for water retention based on underlying geology and the structural integrity of the bog. Snowpack also provides a water source throughout the spring and summer thaw (Wimbush 1970; Whinam et al. 2010).

Sphagnum moss growth and peat accumulation

Sphagnum moss is an integral component of the ecosystem type (Figure 78). It grows 0.9-7.3 cm in depth/year (Whinam & Buxton 1997) and its presence is limited by minimum summer inflow (Wimbush 1970) and evapotranspiration in the driest month (Whinam et al. 2003). *Sphagnum* contributes considerably to peat formation (Hope et al. 2009, 2012), which occurs when primary production exceeds the decomposition of vegetation. The rate of peat accumulation varies from 0.1-10 cm/100 years in montane bogs (Hope & Whinam 2005), and 1-3 cm/100 years in higher altitude bogs (Hope et al. 2012). Over thousands of years, peat can become several metres deep, creating an important carbon store (Hope et al. 2012). Peat and *Sphagnum* have significant water holding capacity, which promote positive feedback on waterlogging. They may also be important in modulating water flow and maintaining the hydrology of surrounding vegetation (Ashton & Williams 1989), although at a local level rather than a regional scale (Western et al. 2008) and over relatively short timeframes of days rather than weeks (Wimbush 1970).

Fire regimes

Unlike peatlands in most other parts of the world, Australian *Alpine Sphagnum Bog and Associated Fen* occur in naturally fire-prone climates and landscapes. Fires ignited by lightning in mountainous terrain may spread if not followed by drenching rain. Fires may also spread upslope from adjoining lowland landscapes from lightning or human ignitions. Fuel moisture varies seasonally and interannually and may be sufficient for fire spread through bogs during late summers of some years. Climate change is increasing the occurrence and severity of both drought and extreme fire weather. Due to their long history of evolution in a fire-prone environment, many characteristic species have traits that enable persistence through certain fire regimes. These include subterranean organs that permit survival and vegetative recovery, fire-cued soil seed banks and behavioural traits that enable early detection and retreat to refuges such as burrows (e.g., Camac et al. 2017). *Sphagnum* has no recovery organs or seed bank, but may partially survive fire despite scorch in some scenarios due to its high moisture retention, especially in basal clefts between hummocks where penetration of lethal heat may be limited (Whinam et al. 2003).

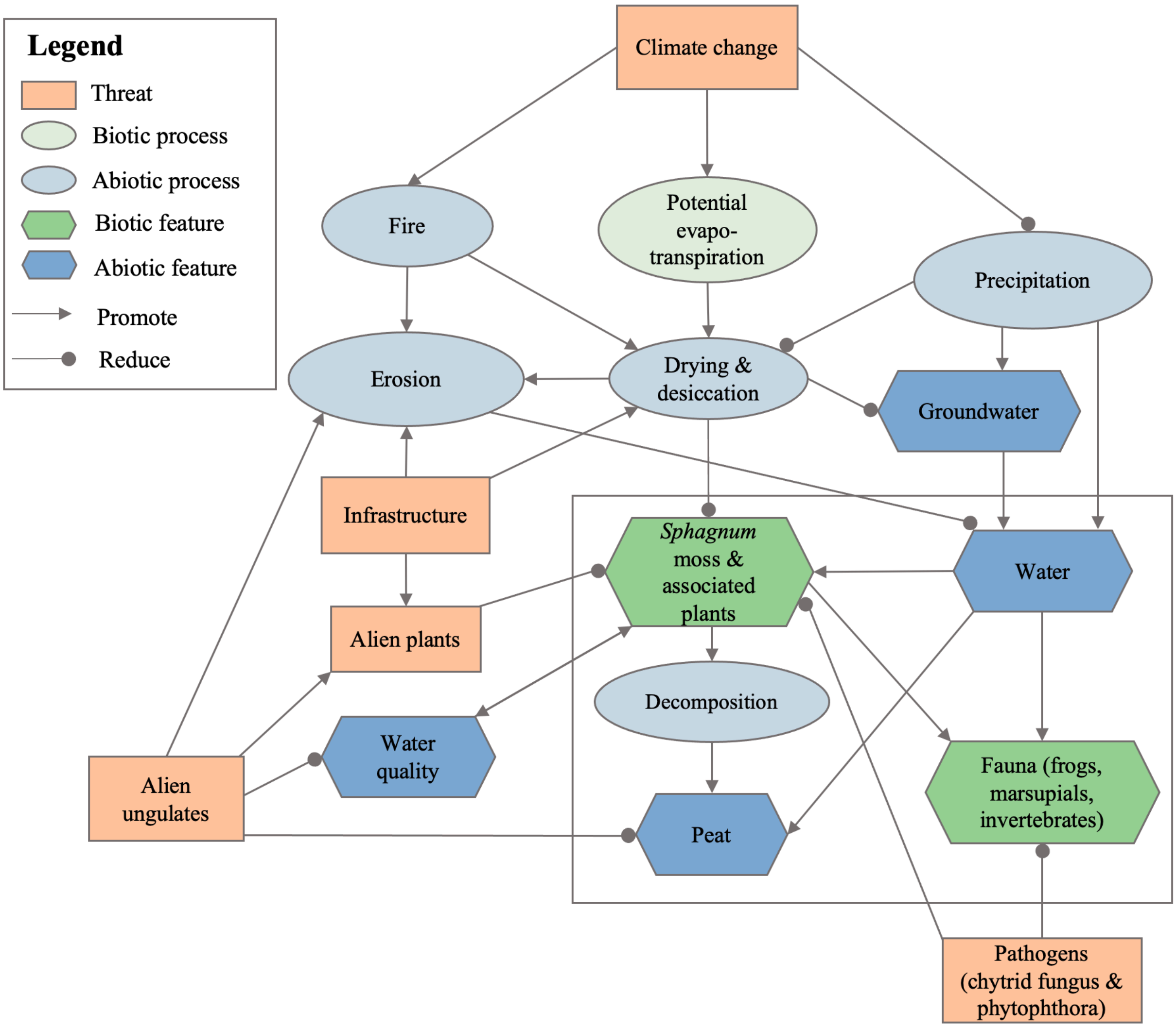


Figure 78. Conceptual model for *Alpine Sphagnum Bog and Associated Fen* showing key components as hexagons (green – biotic, blue – abiotic), ecological processes (ellipses), threats (rectangles) and their positive (arrows) and negative (circles) interrelationships.

Major threats

Climate change

Climate change poses major threats to *Alpine Sphagnum Bog and Associated Fen* both on the mainland and in Tasmania (Clarke & Martin 1999; Whinam et al. 2003; Whinam & Copson 2006; McDougall & Walsh 2007; White 2009) by affecting bog hydrology, peat development and by exacerbating other threats (Figure 78). Increasing temperatures and altered precipitation regimes and reduced snowfall, increased snowmelt regimes and thus altered groundwater movement. Under these conditions, higher rates of desiccation may limit *Sphagnum* cover and growth (Wimbush 1970) as water requirements increase, affecting the balance between growth and decomposition that underpins peat formation. Drying and shrinking may lead to erosion and structural collapse of parts of the bog and allow dryland plant species to encroach. Many alpine species are expected to decline and may become locally extinct as their habitat becomes unsuitable as the snowline and temperatures ascend the altitudinal gradient (Bennett et al. 1991; Brereton et al. 1995; Green & Pickering 2002)

Shifting fire regimes

Although fire is a natural, historically infrequent process in Australian alpine environments (Banks 1989; Richards et al. 2001; Zylstra 2006), two main factors limit persistence of *Alpine Sphagnum Bog and Associated Fen* through successive fires, both on the mainland and in Tasmania. First, recovery rates are delayed and slow due to short growing seasons and low temperatures (Clarke et al. 2015). Hence short fire intervals may disrupt reproduction, replenishment of seed banks or recovery of individuals, causing populations declines or local extinction of some species. Critical fire intervals vary with context, but could extend to several decades. Second, prolonged dry spells may reduce peat moisture content sufficiently to permit peat combustion if ignitions occur (Whinam & Chilcott 2002; Hope et al. 2005; Prior et al. 2020). Substrate combustion causes mortality of both seed banks and subterranean recovery organs, initiating a very slow recovery process beginning with colonisation of 'brown' mosses, opportunistic recruitment of short-lived forbs and fecund shrubs, and eventual re-entry of *Sphagnum* and peat development that may take many decades or centuries. Climate change is predicted to increase fire frequency and severity, with increased temperatures and reduced rainfall leading to a higher number of days of very high or extreme fire danger (Hennessy et al. 2005; Hennessy 2007; BOM & CSIRO 2018). *Sphagnum* and peat are very fire sensitive and can be destroyed by severe fires (Figure 79) (Walsh & Mcdougall 2004); *Sphagnum* relies on unburnt fragments to regenerate (Whinam et al. 2010). Shrubs such as *Epacris gunnii* and *Richea continentis* are less sensitive to fires as they can recolonise from seeds and there is potential positive feedback between shrub establishment and fire (Camac et al. 2017). However, germination may take up to eighteen months (Whinam et al. 2010) and delays in germination can limit post-fire recovery as bare soils are vulnerable to soil erosion, ungulate trampling and weed invasion (Walsh & Mcdougall 2004; McDougall 2007). Repeated burning at short intervals and high severity are likely to be highly detrimental to *Alpine Sphagnum Bog and Associated Fen,* particularly under drought conditions (Whinam et al. 2010). Recovery of pre-fire vegetation structure after a single fire requires many decades (Walsh & Mcdougall 2004; Good 2006; McDougall 2007), and peat accumulation may require centuries, if it can recover at all.

Alien ungulates

Introduced ungulates (cattle, feral horses, deer and pigs) have substantially degraded *Alpine Sphagnum Bog and Associated Fen* (Figure 79). Activity by ungulates can dislodge or overturn bog vegetation, trample large areas (sometimes down to rocky pavement), create wallows, tracks and drainage lines, foul waterways, damage fauna habitat, change species composition and structure, hinder post-fire recovery, and spread weeds (Costin 1954; McDougall 1982, 2007; Kauffman & Krueger 1984; Lawrence 1995; Whinam & Comfort 1996; Williams et al. 1997; Wahren et al. 2001; Whinam & Chilcott 2002; Whinam et al. 2003; NPWS 2007; Prober & Thiele 2007; White 2009; Cherubin et al. 2019; Driscoll et al. 2019; Robertson et al. 2019). Domestic livestock have now been removed from most *Alpine Sphagnum Bog and Associated Fen*, except cattle in some state forests. Feral horses are of most concern on the mainland, as populations in the alpine region of NSW increased from ~9,000 in 2014 to > 25,000 in 2019 (Cairns 2019), and with a major control effort were reduced to 17,393 (range 12,797 - 21,760) in 2023, with 9,000 horses removed or culled since November 2021. Control efforts are ongoing, with the aim to reduce numbers to below 3000 by 2027. Even a few horses are unsustainable, as damage to *Alpine Sphagnum Bog and Associated Fen* and *Alpine-subalpine Streams* is cumulative (Tolsma & Shannon 2018). Feral deer and pigs pose similar threats to integrity of bog soils and vegetation on the mainland and, to a lesser extent, in Tasmania. Grazing by native marsupials (mostly in Tasmania) can also modify the vegetation composition and structure.

Alien plants

Alpine Sphagnum Bog and Associated Fen degraded by fire, grazing or trampling is highly susceptible to weed incursions, while intact bogs display some resilience to invasion (Wimbush &

Costin 1979). High abundance of *Juncus effusus* (Soft rush) has permanently altered the structural integrity of some *Alpine Sphagnum Bog and Associated Fen* (Walsh & Mcdougall 2004; McDougall 2007). Other common weeds in mainland bogs are a result of a long prior history of cattle grazing, such as *Holcus lanatus* (Yorkshire fog), *Anthoxanthum odoratum* (Sweet vernal-grass) and *Phleum pratense* (Timothy grass) (McDougall & Walsh 2007; Tolsma 2009). *Salix* spp. (Willows) are widespread in some mainland alpine areas (particularly Victoria since the 2003 fires) (Tolsma & Shannon 2007; McMahon et al. 2009), with *Salix cinerea* (Grey sallow willow) of most concern (NHT 2003). The deep root systems, high water use, and cold tolerance gives willows the capacity to alter the vegetation structure and hydrology of upland riparian and wetland systems (Cremer 1995).

Soil erosion

Fire, trampling, and grazing can cause soil erosion (e.g., channelling, tunnelling, or exposure of rocky pavement), which may reduce or destroy the water holding capacity of *Alpine Sphagnum Bog and Associated Fen*, with impacts well documented in mainland bogs (Wimbush & Costin 1983; Ashton & Williams 1989; Wahren et al. 1999; White 2009). Changes in water retention and water quality can reduce the filtration of nutrients (e.g., nitrates and sulphates; Silvester 2006, 2007) and reduce the supply of organic carbon or buffering capacity, which are important for downstream ecological processes. Sedimentation also occurs in bogs when overgrazing occurs in their catchments, and many bogs bear legacies of past sedimentation initiated by historical grazing practices (Wimbush & Costin 1983).

Developments

Infrastructure development (e.g., hydroelectric schemes or aqueducts) can degrade *Alpine Sphagnum Bog and Associated Fen* by altering the structure or hydrological processes, complete inundation, or draining the water supply (Lawrence 1999, 2001; McDougall & Walsh 2007). Ski resorts, carparks, and roads can directly damage *Alpine Sphagnum Bog and Associated Fen* or alter its hydrology, by interfering with water flow. Recreational use can cause trampling of vegetation (Whinam & Chilcott 2002; McDougall & Walsh 2007) or off-road vehicle impacts, especially in the summer when visitation is higher.

Pathogens

The plant pathogen *Phytophthora gregata* has recently been found in *Alpine Sphagnum Bog and Associated Fen* in Kosciusko National Park, NSW, and is known to affect several shrub species (Rigg et al. 2018). For example, *Pimelea bracteata* (Rice flower) has been listed as Critically Endangered due to a rapid decline from this pathogen (TSSC Preliminary Determination, 15 November 2019). Frogs are also at risk from Amphibian Chytrid Fungus (Clemann et al. 2009).

Figure 79. Left: Burnt bog at Jacks Creek, Tasmania, 2016 (Photo: Kathryn Storey). Right: Impacts of threats to *Alpine Sphagnum Bog and Associated Fen* near Cowombat Flat, Victoria, trampled to mud by feral horses (Photo: Arn Tolsma).

IUCN Stresses Classification

1.2 – Ecosystem degradation
1.3 – Indirect ecosystem effects

IUCN Threats Classification

1.3 – Tourism & Recreation Areas
2.3 – Livestock Farming & Ranching
6.1 – Recreational Activities
7.1.1 – Increase in Fire Frequency/Intensity
7.2 – Dams & Water Management/Use
8.1 – Invasive Non-native/Alien Species
8.1.2 – Named Species
9.1.2 – Run-off
11.2 – Droughts
11.3 – Temperature Extremes

Ecosystem collapse

Alpine Sphagnum Bog and Associated Fen is collapsed when it has lost either its characteristic native biota or its peat or water holding hydrological properties.

The biotic indicators of collapse represent decline of Sphagnum moss and/or other characteristic alpine bog vascular flora. First, *Sphagnum* is the major diagnostic species for *Alpine Sphagnum Bog and Associated Fen*. It requires a favourable moisture regime to establish, but can then act as an ecosystem engineer, depressing the growth of vascular plants and setting up a positive feedback to its own growth and ultimately peat accumulation (van Breemen 1995) as well as resistance to combustion due to its moisture retention capacity. Sphagnum moss may be eliminated by severe fires, Sphagnum harvesting, hydrological disruption to surface or ground water flows, or trampling/digging activity by introduced herbivores. Even when fires burn into *Alpine Sphagnum Bog and Associated Fen,* the ecosystem may recover if at least some *Sphagnum* fragments remain alive, and moisture supply is sufficient for vegetative expansion (Clarke et al. 2015). The combination of changes in fire and moisture availability can promote invasion of other shrubs and/or sedges (including buttongrass in Tasmania) that outcompete *Sphagnum*.

Secondly, *Alpine Sphagnum Bog and Associated Fen* can be considered collapsed when other functionally important characteristic biota fall below a critical level of abundance. These states can be recognised by the lack of groups of indicative vascular plant species: restiads or cordrushes, *Empodisma minus* and *Baloskion australe* throughout both the mainland and Tasmanian bogs; and characteristic bog shrubs, *Epacris* spp. *(E. paludosa, E. glacialis, E. celata, E. breviflora* or *E. gunnii* on the mainland, or *E. gunnii* or *E. serpyllifolia* in Tasmania*), Richea* spp *(R. continentis* on the mainland, or *R. victoriana, R. gunnii, R. scoparia* in Tasmania*), Baeckea* spp. *(B. gunniana* throughout, or *B. latifolia* or *B. utilis* on the mainland*).*

Potential abiotic indicators of collapse include the extent and depth of peat (Carroll et al. 2023), water table depth or soil moisture (Mason et al. 2021), extent and frequency of severe fires and substrate fires (Good et al. 2010; Whinam et al. 2010) and or climatic aridity index (Zomer et al. 2023), which measures the balance between precipitation and evapotranspiration. Some piezometric monitoring of water table depth has been carried out for bogs within the Snowy Hydo-electric Scheme in NSW, but may not provide sufficient coverage for analysis. Fire history data are available from at least c. 1960 to present. Time series of aridity index was calculated historically and as projections into the future.

Characteristics of the collapsed state will vary (Figure 79), depending on the threats driving decline and the speed at which they act, as well as the initial characteristics of the bog such as geology, topographic position, and floristic composition (both within and outside the bog). The collapsed state can sometimes be rocky pavement, bare ground mud, or a sward of introduced plant species but is often another recognisable alpine vegetation community, including:

- Wet tussock/sod grassland (from hummock-dominated bogs toward valley bottoms or on exposed slopes where soils are deeper and cold air drainage suppresses shrub growth) – classified as (derived or degraded) *Alpine-subalpine Damp Valley Grassland and Rushland*
- *Tussock grassland* (on geographic saddles after fire, observed in ACT; Whinam pers. comm.)
- Sedgeland (from bogs on valley bottoms with poor drainage) –classified as (derived or degraded) *Alpine-subalpine Damp Valley Grassland and Rushland*
- Shrubland (usually from bogs on shallower soils on slopes, where they were previously dominated by shrubs) – these may transition autogenically to *Alpine-subalpine Closed heath* or *Alpine-subalpine Open Grassy Heathland and Herbfield.*

In summary, ecosystem collapse occurs when:

1. *Area*: The mapped distribution declines to zero (100% loss) (Criteria A and B), and/or;
2. *Moisture balance* declines below the range of variation represented within the current distribution of the bogs (Criterion C), and/or;
3. *Fire* occurs with drought in the same season *(Criterion C), and/or;*
4. *Sphagnum,* restiads or bog shrubs fall below a critical level of prevalence, e.g., < 5% (Criterion D).

Risk Assessment

Criterion A: Reduction in distribution

Summary

Under criterion A, the risk status of *Alpine Sphagnum Bog and Associated Fen* is **Endangered (Endangered-Critically Endangered)** on the mainland, **Least Concern** in Tasmania, and **Vulnerable (Least Concern to Vulnerable)** in Australia, as a consequence of historical declines assessed under sub-criterion A3.

Methods

There is wide agreement among experts that the ecosystem type has significantly reduced in distribution since European colonisation (Costin 1954, 1957; Wimbush & Costin 1983; Ashton & Williams 1989; Lawrence 1999). However, losses have not been quantified nor mapped, as livestock grazing predate ecological monitoring by almost 100 years (McDougall 1989). Losses were therefore estimated from indirect measures of decline reported in the literature.

Assessment outcome

Recent change (A1)

On mainland Australia, repeated fires, their interaction with drought and exposure to wild horses and deer have caused degradation of bogs in some regions, leading to reductions in bog distribution (Wimbush & Costin 1983; Ashton & Williams 1989; Walsh & Mcdougall 2004). *Sphagnum* is likely to recover at least in part where fire was not severe or not followed by drought (Camac 2014; Clarke et al. 2015). On the mainland, severe wildfire in 2003, following a protracted drought, burnt almost the entire mainland distribution of alpine bogs, of which ~14 (12-16)% was estimated to have collapsed (Table 31) due to consumption of peat and *Sphagnum* cover, resulting in transformation of hydrological function and biota (Tolsma et al. 2005; Good et al. 2010). A similar combination of fire and drought occurred in 2020, which burnt 32% of the area of mainland bogs (Keith et al. 2022a). Assuming a similar fraction of peat consumption, a further 4 (4-5)% of bog area may have collapsed in 2020 (Table 2). Fires also affected smaller parts of the mainland distribution of bogs in 1998, 2006/7 and 2009, and, although no estimates of peat consumption are available for these events a similar fraction of peat consumption estimated for the 2003 fire, could account for a further 2 (1-3)% reduction in the mainland distribution. Combining these estimates suggests a total reduction in mainland bog over the past 50 years of 20 (17-24)%, resulting in **Least Concern** status under Sub-criterion A1.

Table 31. Estimates of decline in distribution of *Alpine Sphagnum Bog and Associated Fen* attributable to fires over the past ~50 years following an era of major livestock grazing impacts.

	Proportion of distribution burnt			**Proportion of distribution in which fire consumed peat/Sphagnum**		
Fire year	***Mainland***	***Tasmania***	***National***	***Mainland***	***Tasmania***	***National***
since 1960	~100%	16%	5%		1%	<1%
1998	no estimate	0	no estimate	no estimate	no estimate	
2003	>90%	0	65%	14(12-16)%		11(9-13)%
2006/7	no estimate	0	no estimate	no estimate	no estimate	
2009	no estimate	0	no estimate	no estimate	no estimate	
2015-18	~0	3	1			1%
2020	32%	0	21%	4(4-5)%		3(2-4)%
Overall decline since 1960 excluding fires in 1998, 2006/7 & 2009:						*15(12-18)%*
Overall decline since 1960 based on best estimate above and assuming 2% area of peat loss in fires of 1998, 2006/7 & 2009						17%
Overall decline since 1960 based on lower bound above and assuming 1% area of peat loss in fires of 1998, 2006/7 & 2009						13%
Overall decline since 1960 based on upper bound above and assuming 3% area of peat loss in fires of 1998, 2006/7 & 2009						21%

In Tasmania, ~16% of bogs have likely been burnt at least once since 1960/61, with 2.46% burnt in 2016 ((Bowman et al. 2021, although estimates of peat consumption are unavailable. *Sphagnum* moss harvesting from Tasmania is estimated to have resulted in a reduction of less than 1% of the ecosystem type's distribution (Whinam et al. 2003). If similar rates of peat consumption estimated for the 2003 mainland fire are assumed for Tasmania, a decline of 3(2-4)% in the distribution of *Alpine Sphagnum Bog and Associated Fen in Tasmania* is likely to have occurred over the past 50 years. Its risk status is Least Concern.

Combining these estimates over the national range suggests an overall reduction in distribution of 17 (13-21)% (Table 31). Therefore, the status of *Alpine Sphagnum Bog and Associated Fen in Australia* is **Least Concern** under Sub-criterion A1.

Future change (A2)

Future declines in *Alpine Sphagnum Bog and Associated Fen* are likely due to climate change via decreased water availability, increasing incidence of severe drought, exposing peat to combustion, and increasing incidence of high fire danger days (Hennessy et al. 2005; Hennessy 2007; BOM & CSIRO 2018). These future declines are likely to be compounded by damage from trampling by wild deer and horses, if populations of those introduced ungulates are not adequately controlled (Tolsma & Shannon 2018). Few quantitative projections of alpine bog distribution under future climates are available. White (2009) modelled the distribution of alpine bogs distribution within the 10.6 km^2 region of the Bogong High Plains, Victoria. She projected hydrologically suitable habitat would decline in extent by 7.5% to 63% by 2070, under optimistic wet climate scenario and a pessimistic dry climate scenario, respectively. Similar scenarios of decline may be expected in other parts of the distribution of mainland alpine bogs due to similar regional climate trends in mountainous areas of southeastern Australia. Fire-related reductions may be expected to be greater in the next 50 years than in the past 50 years (see Sub-criterion A1, Table 31). However, further work is needed to quantify projected climate-driven declines in distribution, as well as declines attributable to other causes such as construction of energy infrastructure, fire-related processes and habitat transformation by feral pigs, deer and horses. The risk status is therefore currently **Data Deficient** under Sub-criterion A2.

Historical change (A3)

Historical changes in distribution of Alpine Sphagnum Bog and Associated Fen may be inferred by aggregating estimates of decline attributable to multiple causal agents. Here, we take estimates of reduction in distribution attributable to severe fires in the past 50 years (see Sub-criterion A1, Table 31) and combine them with estimates of earlier reductions attributable to other causes.

Historical reductions attributable to livestock grazing

Heavy grazing by livestock occurred in mainland alpine areas from the mid 19th until the mid 20th century (Lawrence 1999). Ungulate activity removed native vegetation, created bare ground, and pugged soils, transformed *Alpine Sphagnum Bog and Associated Fen* transformed to short-grazed turf. On the Nunniong Plateau (33 bogs, Tolsma 2008) and Alpine National Park (65 bogs, Tolsma 2008b), the reduction in functional bog area varied 2-95% between individual bogs (mean 29.3%, 95% confidence interval: 23.5-35.1%). Several other Victorian mountains that previously contained *Alpine Sphagnum Bog and Associated Fen* are now considered 'dry' and thus devoid of the ecosystem type (e.g., Mount Feathertop, Loch, Hotham and Howitt; Costin 1957; Lawrence 1999), including around Mounts Nelse and Bogong, and Ruined Castle (Lawrence 1999). An updated review of this estimate, based on a wide range of data, anecdotal evidence and the site survey data proposed a different estimate of historical loss (Tolsma pers. comm.). With an extant area of 4486 ha calculated from GIS mapping, the area lost since 1750 is estimated to be around 19.4% (calculated as area lost divided by the sum of extant area and area lost: 1080 / (1080 + 4486)) with lowest estimate of 13.9 % and highest of 22.9% (Table 32).

Table 32. Total loss of bog in Victoria since 1750 distributed across tenure and regions.

Region	**Tenure**	**Area bog (ha, 2009)**	**Ha lost best**	**Ha lost min**	**Ha lost max**
Baw Baw	Alpine Resorts	19	3	1	5
Baw Baw	Parks & Reserves	543	3	1	5
Baw Baw	State Forest	140	10	2	15
Bogong HP, Mt Hotham	Freehold	71	35	20	50
Bogong HP, Mt Hotham	Parks & Reserves	1630	347	280	400
Bogong HP, Mt Hotham	State Forest	180	50	30	70
Falls Creek	Alpine Resorts	50	15	10	20
Lake Mountain	Parks & Reserves	33	5	1	8
Mt Buffalo	Parks & Reserves	133	40	30	50
Mt Hotham	Alpine Resorts	4	2	1	4
Nunniong Plateau / Davies Plain	Parks & Reserves	560	250	150	300
Nunniong Plateau / Davies Plain	State Forest	315	150	100	180

Snowy Range / Moroka	Freehold	56	30	20	50
Snowy Range / Moroka	Parks & Reserves	660	120	70	150
Snowy Range / Moroka	State Forest	92	20	10	30
	TOTAL	**4486**	**1080**	**726**	**1337**

These estimated areas of bog lost reflect losses in regions with known extant occurrences and do not account for loss of entire bogs from other localities, or changes since 2009. Conservatively, we estimate that additional but unquantified losses have occurred and round up the estimated decline in distribution of alpine bogs in Victoria from ungulate grazing since 1750 as 20% (15-25%).

In NSW, extensive tracts of *Alpine Sphagnum Bog and Associated Fen* had been lost or partially lost by the 1950s (Costin 1954). Estimates from the 1950s suggested that bog area had declined by ~50% across the alps in NSW, Victoria and the Australian Capital Territory, during the grazing era (early 1800s to the late 1900s) as a result of heavy grazing of the mire vegetation and physical destruction of the peatbeds by cattle and sheep trampling (Costin 1952). However current opinion of experts suggest that the estimated losses from grazing are more likely to be a similar level to that in Victoria (pers. comm., Keith McDougall), i.e. 20% (15-25%) since 1750.

Concern about land practices led to incremental withdrawal of grazing licences from national parks in NSW between 1944 and 1961 (Wimbush and costin 1979a), from the ACT c.1910 (pers. comm. Keith McDougall), and from Victoria between the 1950s and 2006 (DELWP 2015). Aerial photos show little recovery in bog area since 1944 at 15 sites in northern Kosciuszko (Keith McDougall, unpublished data), although small increases in wetland vegetation cover were observed on the Bogong High Plains, Victoria, between 1936 and 1980 (McDougall 2003) following declines in stocking rate.

In Tasmania, sheep grazing increased the cover of bare ground across the Central Plateau in general (Bridle & Kirkpatrick 1999). However, sheep grazing only affected a small area of *Alpine Sphagnum Bog and Associated Fen* (Jennie Whinam, pers. comm.) and ceased on the Central Plateau of Tasmania in the early 1990s. We assume a negligible reduction in the distribution of Tasmanian bogs because of grazing.

Given that the mainland contributes two-thirds of the extant area of *Alpine Sphagnum Bog and Associated Fen in Australia*, overall losses of the ecosystem type due to grazing are estimated to be lower than 20%.

Historical reductions attributable to fire

Estimates of reductions attributable to peat and *Sphagnum* consumption by fire are given under Sub-criterion A1. Additional historical reductions are likely to have occurred as a result of extensive severe fires in mainland alpine regions in 1926, 1939 and 1951/52 (Zylstra 2006). Almost all mainland bogs have likely burnt at least once since 1939, and in Tasmania, ~16% of mapped bogs have likely been burnt at least once since 1960/61, particularly in 2016 (Prior et al. 2022). The reductions attributable to these historical fires and others undocumented in the 19th and early 20th centuries, could be of a similar magnitude to reductions attributable to fires in the last 50 years. Conservatively, and allowing for some recovery of *Sphagnum* cover where peat was not consumed, we estimated reductions of a further 5 (1-8)% on the mainland, 2 (1-3)% in Tasmania, and 4 (1-6)% in Australia due to fires in the post-industrial era prior to the last 50 years.

Historical reductions attributable to infrastructure development

In Victoria, 125 ha or 1-2% of the pre-1750 area of *Alpine Sphagnum Bog and Associated Fen* was lost when the Kiewa Hydro-Electric Scheme (Lawrence 1999) flooded in the 1940s. A further 92 ha of Alpine Bog in the Bogong High Plains was lost when 35 km of aqueducts were installed (Tolsma 2008b) and areas above the aqueduct are likely to be compromised by increased drainage, with an assumed loss of ~50% of individual bog area. Thus, we estimate that aqueducts on the Bogong High Plains have led to the loss of ~113 ha of *Alpine Sphagnum Bog and Associated Fen* (~1.5% of national pre-1750 area). Losses in NSW, especially in the Snowy Mountains, are assumed to be similar or larger, given infrastructure associated with ski resorts, roads, fire trails, the Snowy Hydro Scheme and Snowy Hydro 2.0. Thus, reductions in distribution of mainland bogs attributable to infrastructure amount to 3% (2-4%) of the pre-industrial historical distribution.

Very small areas of *Alpine Sphagnum Bog and Associated Fen* in Tasmania have been impacted by infrastructure, mainly on the Central Plateau (Jennie Whinam, pers. comm.).

Combining estimates for the Mainland and Tasmania, we estimate the reduction in distribution of *Alpine Sphagnum Bog and Associated Fen in Tasmania* attributable to infrastructure development is 2% (1-3%) of the national pre-1750 distribution.

Loss from sphagnum extraction

Harvesting for the horticultural industry has had minor impacts on *Alpine Sphagnum Bog and Associated Fen*, mostly at lower elevations. The *Sphagnum* moss harvesting industry in Australia expanded in the 1970s, with exports levelling off in the early 1990s at ~15 tonnes of moss exported from Tasmania annually, and ~5800 m^3 of peat extracted annually until the early 2000s (Whinam et al. 2003), mostly below 800 m elevation. There have historically been sites harvested on mainland Australia, but only in very small areas and outside the alpine zone (Whinam et al. 2003). *Sphagnum* moss harvesting is now generally restricted to very small areas on private land. Overall historic loss was probably < 1%.

Overall estimate of reduction

Aggregating the declines attributable to different causes as a proportion of pre-1750 extent, as estimated above (Table 33) and assuming no overlap in these estimates (additive effect of each cause of reduction), produces estimates of maximal historical reductions in distribution of *Alpine Sphagnum Bog and Associated Fen* of 48(35-61%) on the mainland, 5(3-7)% in Tasmania and 35(26-45%) in Australia. Based on a precautionary, but realistic interpretation of uncertainty represented in these bounded estimates under sub-criterion A3, the risk status of *Alpine Sphagnum Bog and Associated Fen on mainland Australia* is **Least Concern (Least Concern to Vulnerable),** the risk status of *Alpine Sphagnum Bog and Associated Fen in Tasmania* is **Least Concern (Least Concern to Least Concern), and** the risk status of *Alpine Sphagnum Bog and Associated Fen in Australia* is **Least Concern (Least Concern to Near Threatened)**.

Table 33. Estimated reductions since industrialisation (notionally 1750) in distribution of *Alpine Sphagnum Bog and Associated Fen* on the mainland, Tasmania and in Australia attributed by cause. The overall reduction assumes additive effect of all cause and is thus a maximum bound considering likely overlap in causes of reduction.

Cause of reduction	Mainland	Tasmania	Australia
Livestock grazing	20 (15-25)%	< 1%	13 (10-16)%
Severe fires 1975-2025	20 (17-24)%	3 (2-4)%	16 (13-19)%
Severe fires 1750-1975	5 (1-8)%	2 (1-3)%	4 (1-7)%
Infrastructure development	3 (2-4)%	< 1%	2 (2-3)%

Sphagnum extraction	< 1%	< 1%	< 1%
Overall reduction (max)	48 (35-61)%	5 (3-7)%	35 (26-45)%

Criterion B: Restricted distribution

Summary

Based on criterion B, *Alpine Sphagnum Bog and Associated Fen* was found to **Vulnerable** on the mainland (B1 aii; b), **Endangered** in Tasmania (sub-criterion B1 ai+ii; c) and **Least Concern** in Australia (Table 34).

Methods

We used the distribution map that incorporates mapped units verified by experts and adapted for consistency between jurisdictions (see *Criterion A* and main methods) to calculate the Extent of Occurrence (EOO; sub-criterion B1) and Area of Occupancy (AOO; sub-criterion B2).

Most threats to *Alpine Sphagnum Bog and Associated Fen* (introduced plants and ungulates) show localised effects and, despite their substantial long-term effects, are unlikely to trigger severe changes in distribution over the short term. Climate change is also considered a severe threat, but one that acts over an extended period. Therefore, the number of threat-defined locations (sub-criterion B3) was based on fires, which is a widespread and severe threat that can quickly affect large areas of the ecosystem type.

Assessment outcome

Sub-criterion (B1)

The EOO for alpine bogs was calculated as 33,883 km^2 on mainland Australia, 18,511 km^2 in Tasmania, and 146,668 km^2 combined for Australia (Table 34; Figure 80). There is evidence of continuing decline in extent and environmental quality, as well as multiple threatening processes (see sections on criteria A and C and Threats). Under sub-criterion B1, *Alpine Sphagnum Bog and Associated Fen* is therefore **Vulnerable** on mainland Australia, **Endangered** in Tasmania and **Least Concern** in Australia.

Sub-criterion (B2)

The number of 10×10 km grid cells (Area of Occupancy, AOO) occupied by *Alpine Sphagnum Bog and Associated Fen* is 103 on mainland Australia, 38 in Tasmania, and 138 in Australia (Table 34; Figure 80) based on available map data and excluding cells that account for <1% of the total mapped extent in accordance with Red List guidelines (IUCN 2024). There is evidence of continuing decline in extent and environmental quality, as well as multiple threatening processes (see sections on criteria A and C and Threats). Under sub-criterion B2, *Alpine Sphagnum Bog and Associated Fen* is therefore **Least Concern** on mainland Australia, **Vulnerable** in Tasmania and **Least Concern** in Australia.

Sub-criterion (B3)

Fires can spread over large areas but do not affect all mountain ranges simultaneously with combustion of substrate and *Sphagnum*. This is likely due to differences in fire weather at the time of spread, as well as topographic complexity across the ecosystem type's distribution resulting in a wide range of climatic conditions between and within the two major regions (mainland and Tasmania). For example, very severe fires in 2003 and 2019/2020 affected> 20% of the mainland bog distribution. Since 1960, almost the entire mainland distribution has been burnt at least once in major fire events (Table 33). In Tasmania, fire history shows numerous fires since 1960/61, affecting ~16% of bog distribution there (Table 33). Extensive fire events affecting large portions of bog distribution are consequence of regional-scale antecedent weather conditions that affected much of southeastern Australia (Nolan et al. 2016; Abrams et al. 2021). Landscape flammability is influenced heavily by regional weather cycles that may extend through multiple years, such that fires in different years may be only semi-independent. For example, 2 (BOM 2021) and extreme multi-year events are increasing in frequency (Lu et al. 2025). Multiple large fire events occurred on mainland alpine regions within 20 years (2003-4, 2009, 2013, 2020). Nonetheless, geographic separation and topographic variation within the distribution of *Alpine Sphagnum Bog and Associated Fen* suggests that some areas will remain unburnt in major fire events, as they have in previous years. Based on these regional and local influences, we estimate that risks from fire to alpine bogs is likely spread over 5-10 threat-defined locations on mainland Australia, and 4-6 locations in Tasmania, with a total of 9-16 locations across the Australian distribution of alpine bogs and could trigger major changes within the next 20 years. Therefore, under sub-criterion B3, *Alpine Sphagnum Bog and Associated Fen* is **Vulnerable** in Tasmania (plausible range Least Concern to Vulnerable) and **Least Concern** in Australia and mainland Australia (with a plausible range of Least Concern to Vulnerable on the mainland).

Table 34. Summary of metrics and assessment outcomes under criterion B for *Alpine Sphagnum Bog and Associated Fen* on mainland Australia, in Tasmania, and in Australia. AOO estimates exclude marginal occurrences <1% of total extent (IUCN 2024).

Type	**Metric**	**Estimate**	**Criterion**	**Category**	**Rationale**
Mainland	EOO	33883.1	B1aii; b	VU	EOO metric below thresholds for the VU category, and following conditions met. Observed or inferred continuing decline in measure of environmental quality. Threatening processes. Ecosystem type present at 100 threat defined locations.
Mainland	AOO	103	B2	LC	AOO metric well above the thresholds for threatened categories.
Mainland	Location	5-10	B3	LC (LC-VU)	Number of threat-defined locations is marginally within threshold for VU and capable of Collapse or becoming Critically Endangered within a very short period
Tasmania	EOO	18510.6	B1ai+ii; c	EN	EOO metric below thresholds for EN, and following conditions met. Observed or inferred continuing decline in measure of extent. Observed or inferred continuing decline in measure of environmental quality. Ecosystem type present at 5 threat defined locations.
Tasmania	AOO	38	B2ai+ii; c	VU	AOO metric below thresholds for VU, and following conditions met. Observed or inferred continuing decline in measure of extent. Observed or inferred continuing decline in measure of environmental quality. Ecosystem type present at 5 threat defined locations.
Tasmania	Location	4-6	B3	VU (LC-VU)	Number of threat-defined locations is within threshold for VU and capable of Collapse or becoming Critically Endangered within a very short period
Australia	EOO	146667.6	B1	LC	EOO metric well above the thresholds for threatened categories.
Australia	AOO	138	B2	LC	AOO metric well above the thresholds for threatened categories.
Australia	Location	9-16	B3	LC (LC-LC)	Number of threat-defined locations is above threshold for VU and capable of Collapse or becoming Critically Endangered within a very short period

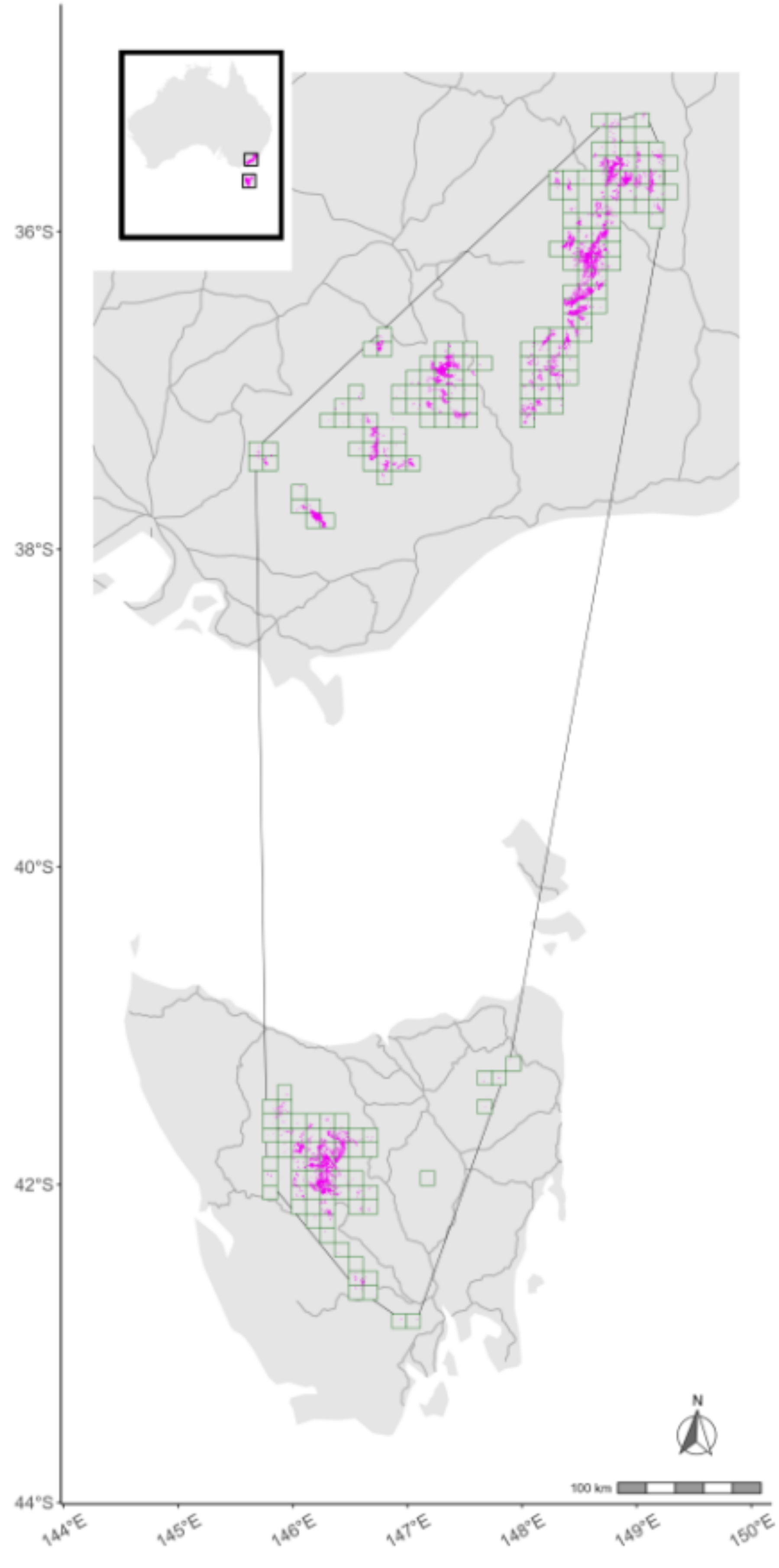


Figure 80. Map of Australian *Alpine Sphagnum Bog and Associated Fen* (magenta polygons), showing EOO (black polygon) and AOO where the 1% rule was not applied (green squares).

Criterion C: Environmental degradation

Summary

We assessed environmental degradation of *Alpine Sphagnum Bogs and Associated Fen* based on four indicators: moisture balance, co-occurrence of fire and drought, drought duration (number of dry months and fire frequency. Statistical models of drought duration and fire frequency were uninformative, and yielded Data Deficient assessment outcomes (Table 35). A pilot analysis projecting the likelihood of fire and drought co-occurrence produced Least Concern status under Sub-criterion C2b. Analysis of Moisture balance over the past 50 years under Sub-criterion C1 indicated **Near Threatened** status (plausible bounds Least Concern-Vulnerable) for mainland bogs, and **Least Concern status** (plausible bounds Least Concern-Vulnerable) for Tasmanian bogs and bogs in Australia. Analysis of moisture balance over future (Sub-criterion C2) and historical (Sub-criterion C3) timeframes produced lower risk results than for the past 50 years. The overall outcomes of assessment for criterion C were therefore the same as reported for Sub-criterion C1.

Table 35. Summary of assessment outcomes for four abiotic indicators under criterion C for *Alpine Sphagnum Bogs and Associated Fen* on mainland Australia, in Tasmania and in Australia.

Indicator	**Ecosystem types**	**C1**	**C2a&b**	**C3**	**Overall C**
Moisture balance (Aridity Index)					
	Mainland	NT (LC-EN)	LC (LC-VU)	LC (LC-LC)	NT (LC-EN)
	Tasmania	LC (LC-VU)	LC (LC-NT)	LC (LC-VU)	LC (LC-VU)
	Australia	LC (LC-VU)	LC (LC-NT)	LC (LC-LC)	LC (LC-VU)
Fire and drought co-occurrence					
	Mainland	DD	LC (LC-LC)	DD	LC (LC-LC)
	Tasmania	DD	LC (LC-LC)	DD	LC (LC-LC)
	Australia	DD	LC (LC-LC)	DD	LC (LC-LC)
Number of dry months					
	Mainland	DD	DD	DD	DD
	Tasmania	DD	DD	DD	DD
	Australia	DD	DD	DD	DD
Fire frequency					
	Mainland	DD	DD	DD	DD
	Tasmania	DD	DD	DD	DD
	Australia	DD	DD	DD	DD

Identification of abiotic indicators

We considered four abiotic indicators to assess risk of collapse from environmental degradation:

- *Climatic moisture balance (Aridity Index)*: ratio between climatological mean annual precipitation and mean annual potential evapotranspiration.
- *Number of dry months*: total number of months with negative difference between precipitation and evapotranspiration derived from remote sensors.
- *Fire frequency*: frequency of fires detected by remote sensors.
- *Fire and drought co-occurrence*: Co-occurrence of fire and drought within the same season.

Indicator: Moisture balance

Relevance to ecosystem function

Alpine Sphagnum Bog and Associated Fen occurs in landscapes where low evapotranspiration and high precipitation combine to produce a positive (wetter) moisture balance (Ingram 1983). Conversely, bogs are unlikely to be sustained in climates with appreciable moisture deficits. Moisture balance was therefore used as measure of climatic suitability for bogs.

Data availability and quality

We used a time series of interpolated climatological measurements to calculate the ratio between mean annual precipitation and mean annual potential evapotranspiration, the Aridity Index (UNEP 1992). Higher values of *Aridity Index* indicate positive moisture balance (wetter) and therefore more suitable environmental conditions for bogs (White 2009). Different spatially explicit time series data sets were used to assess trends in Aridity Index of different timeframes. We extracted the data for all mapped polygons of *Alpine Sphagnum Bog and Associated Fen* for the mainland, for Tasmania and for the combined national distribution.

To assess current trends under Sub-criterion C1, we used monthly precipitation and potential evapotranspiration data from CHELSA-W5E5 version 2.1 (Karger et al. 2023) interpolated spatially from weather station data for 1979-2018. We aggregated the monthly values per year and calculated the aridity index from the annual values. For future trends under Sub-criterion C2, we used mean values of the climatological average Global Aridity Index for the baseline period 1970-2000 (present) and projections for 2021-2040 and 2041-2060. We used projections for *Shared Socioeconomic Pathways* (SSPs) 245, 370 and 420 (IPCC 2022) based on the Mean Model Ensembles rather than the individual Global Circulation models. To assess trends over the historical timeframe under Sub-criterion C3, we used monthly interpolated values from 1901 to 2018 from CRU TS v4.03 (Harris et al. 2014). The data for current and future trends were at 30 arc-second (ca. 1km) spatial resolution, whereas historical data were at half degree spatial resolution (equivalent to 2325 – 2486 km^2 in the study region).

Selection of collapse threshold

In a global context, UNEP (1992) define humid environments as having an Aridity Index > 0.65. To define regionally relevant moisture balance thresholds of ecosystem collapse, we intersected spatial data with the distribution of *Alpine Sphagnum Bog and Associated Fen* (mapped during 1990-2002) with Aridity Index spatial data averaged for 1990-2020. We pooled together aridity data for three years representative of temporal variability in conditions at the time mapping was completed: 1999, 2000 and 2001. We identified the 1st 2.5th and 5th percentiles of aridity values in those years in areas of bog occurrence as marginal conditions for bog persistence. These percentiles correspond to aridity index values of 1.133, 1.228 and 1.329, respectively, as plausible thresholds for ecosystem collapse. Modelled values of the Aridity Index for the CRU data used for Sub-criterion C3 differed from those based on CHELSA data (sub-criteria C1 and C2), due to the coarser spatial resolution of CRU. It was necessary to calculate specific thresholds of collapse for the CRU data using the same methods applied to the CELSA data. The corresponding CRU thresholds applied in Sub-criterion C3 were 0.95, 1.08 and 1.18, respectively, for the 1st, 2.5th and 5th percentiles of aridity values.

Calculation of initial and present/future values

Recent change (C1)

To estimate relative severity of decline in climatic suitability over the last 50 years for *Alpine Sphagnum Bog and Associated Fen*, we accounted for three sources of uncertainty in the analytical workflow: 1) the collapse threshold based on different quantiles; 2) annual variability in moisture balance (aridity index); and 3) the incomplete temporal coverage of the time series. We addressed uncertainty in collapse threshold (point 1) by repeating the analysis for different thresholds described in the previous section. To address annual variability and extrapolation of the time series (points 2 and 3), for each mapped polygon of the ecosystem type, we fitted a linear regression to the annual aridity index values over the 40-year period 1979-2018, weighting polygons by their size. We used the regression to calculate the mean and 95% confidence intervals of aridity index for current (2025) and initial (1976) times, spanning a 50-year period, as required for Sub-criterion C1. We calculated the relative severity of decline for all mapped polygons according to (IUCN 2024) and plotted the cumulative degradation function (cumulative extent of degradation against severity of degradation).

Future change (C2b)

To estimate relative severity of decline in climatic suitability over c. 50 years including the present and the future for *Alpine Sphagnum Bog and Associated Fen*, we accounted for three sources of uncertainty in the analytical workflow: 1) the collapse threshold based on different quantiles; 2) the future greenhouse gas emissions and mitigation action; and 3) the available timeframes for climate projections. We addressed uncertainty in collapse threshold (point 1) by repeating the analysis for different thresholds described in the previous section. To address uncertainty in future emissions and mitigation (point 2), we repeated the analysis for three *Shared Socioeconomic Pathways* using Mean Model Ensembles across global climate models (IPCC 2022): SSP245 (high and immediate emissions reduction, energy from renewables and biofuels, new CO_2 removal technologies, and increasing progress towards Sustainable Development); SSP370 (moderate emissions reduction from 2040s, energy from renewables and fossil fuels, no CO_2 removal technologies, and slow progress towards Sustainable Development); and SSP420 (no or very slow emissions reduction, energy from fossil fuels, no CO_2 removal technologies, and slow and unequal progress towards Sustainable Development). To address uncertainty in the timeframes (point 3), we calculated relative severity using aridity index values for 2030 (2021-2040) and 2050 (2041-2060) as lower and upper bounds for the future, with the same initial aridity values for 1985 (1970-2000). We calculated the relative severity of decline for all mapped polygons according to (IUCN 2024) and plotted the cumulate degradation function.

Historical change (C3)

We assessed historical changes in climatic suitability for *Alpine Sphagnum Bog and Associated Fen*, over a timeframe extending from 1901 to 2018. We excluded earlier trends from the risk assessment because prior to 1901 global climate was strongly influenced by the Little Ice Age 1500 - 1890 (Tibby et al. 2018) and industrial emissions are unlikely to have had an appreciable role in climatic forcing. The workflow to assess Sub-criterion C3 over this historical timeframe was similar to that implemented for Sub-criterion C1, except that the regression was fitted to a longer time series and the aridity index for each mapped bog polygon was derived from coarser resolution climate data (Harris et al. 2014).

Calculation of severity and extent

Recent change (C1)

Mainland Australia

Moisture balance declined over the past 50 years, although for the best and most conservative trend estimates a relatively small portion of the extent of mainland bogs were affected by moderately severe declines in moisture balance (Figure 81). The most precautionary estimates of the trend, however, indicate that moderate to high declines in moisture balance were widespread across almost all of the mainland extent of bog distribution, intersecting Endangered status for the threshold of collapse based on both the 2.5th and 5th percentiles of bog occurrence (Figure 81). *Tasmanian Alpine Sphagnum Bog and Associated Fen* was therefore assessed as **Near Threatened** (plausible range Least Concern – Endangered) under sub-criterion C1.

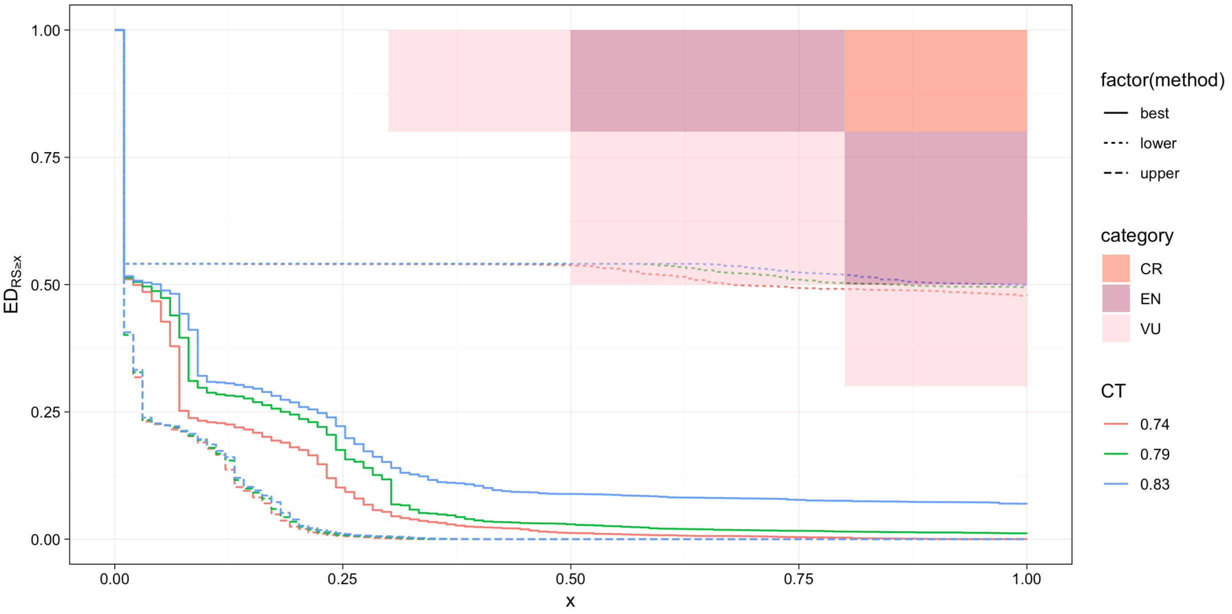


Figure 81. Cumulative degradation function for recent trends in climatic moisture balance (Aridity Index, AI) for *Australian Mainland Alpine Sphagnum Bog and Associated Fen* with the best estimate and 95% lower and upper confidence intervals of the time series regressions for three plausible collapse thresholds (CT 0.74 – 0.83, based on 1st, 2.5th and 5th percentiles of AI across the distribution at the time of mapping).

Tasmania

Moisture balance declined over the past 50 years, although large severe declines occurred over a relatively small portion of the extent for the best and most conservative trend estimates (Figure 82). The most precautionary estimates, however, suggest more extensive declines in moisture balance, with the relative severity and extent of degradation intersecting Vulnerable status for the threshold of collapse based on the 5th percentile of bog occurrence (Figure 82). *Tasmanian Alpine Sphagnum Bog and Associated Fen* was therefore assessed as **Least Concern** (plausible range Least Concern – Vulnerable) under sub-criterion C1.

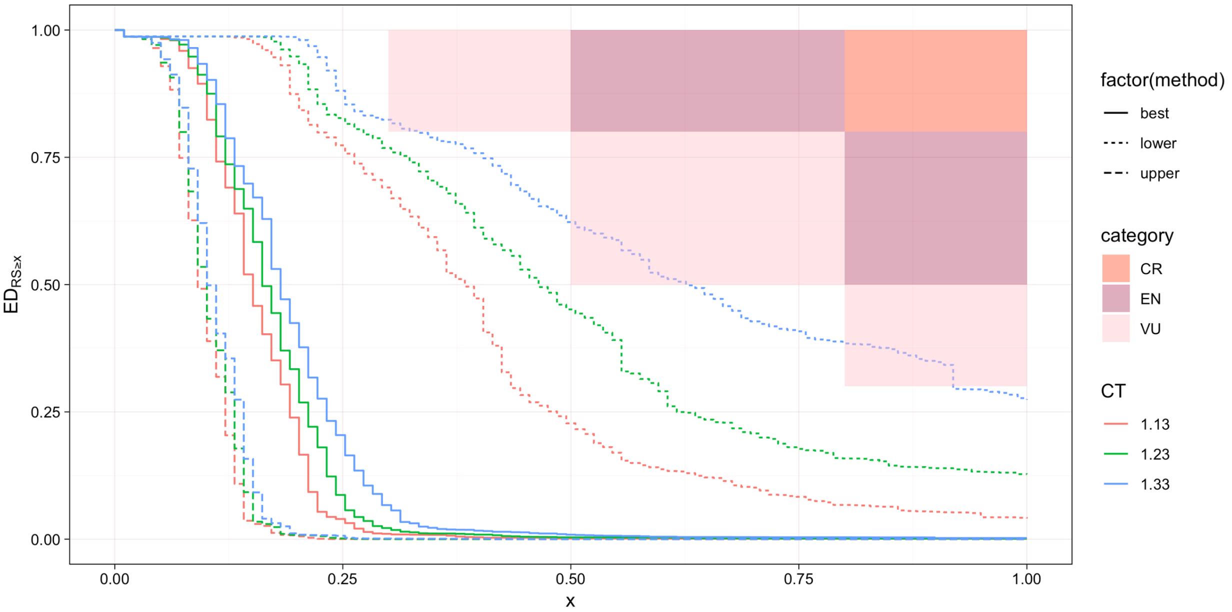


Figure 82. Cumulative degradation function for recent trends in climatic moisture balance (Aridity Index, AI) for *Tasmanian Alpine Sphagnum Bog and Associated Fen* with the best estimate and 95% lower and upper confidence intervals of the time series regressions for three plausible collapse thresholds (CT 1.13 – 1.33, based on 1st, 2.5th and 5th percentiles of AI across the distribution at the time of mapping).

Australia

Moisture balance declined over the past 50 years. Moderate declines occurred over a relatively large portion of the extent for the most conservative trend estimates, meeting the threshold for Vulnerable status (Figure 83). The best and most conservative estimates of trends, however, align with Least Concern status (Figure 83). *Alpine Sphagnum Bog and Associated Fen in Australia* was therefore assessed as **Least Concern** (plausible range Least Concern – Vulnerable) under sub-criterion C1.

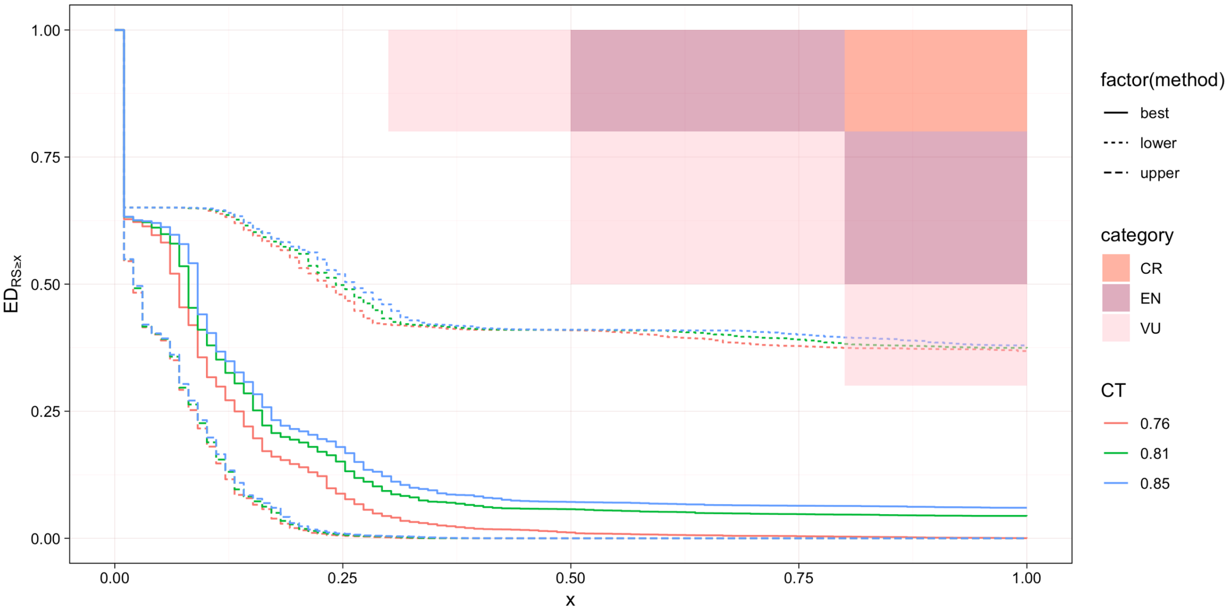


Figure 83. Cumulative degradation function for recent trends in climatic moisture balance (Aridity Index, AI) for *Alpine Sphagnum Bog and Associated Fen in Australia* with the best estimate and 95% lower and upper confidence intervals of the time series regressions for three plausible collapse

thresholds (CT 0.76 – 0.85, based on 1st, 2.5th and 5th percentiles of AI across the distribution at the time of mapping).

Future change (C2b)

Mainland Australia

Most of the scenarios analysed resulted in estimated declines in moisture balance that align with Least Concern status (Figure 84). However, all three SSP scenarios over the longer timeframe for the most precautionary threshold of ecosystem collapse produced declines in Aridity Index with Relative Severity of 30-33% across 80% of the extent, exceeding the threshold for VU status). *Australian mainland Alpine Sphagnum Bog and Associated Fen* was therefore assessed as **Least Concern** (plausible range Least Concern – Vulnerable) under sub-criterion C2b.

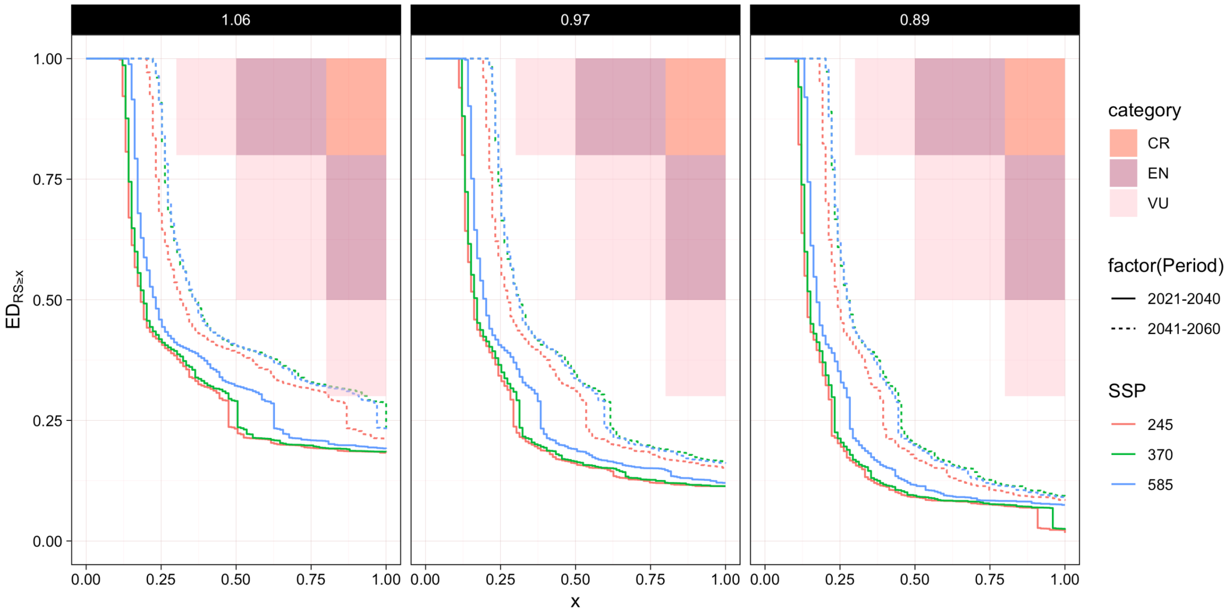


Figure 84. Cumulative degradation function for projected future trends in climatic moisture balance (Aridity Index, AI) for *Mainland Australian Alpine Sphagnum Bog and Associated Fen* with for three plausible collapse thresholds (panels) for three climate change scenarios (SSPs) and two timeframes 1970-2000 to 2021-2040 and 2041-2060 (periods), respectively.

Tasmania

None of the scenarios analysed resulted in estimated declines in moisture balance that met the thresholds for Vulnerable status (Figure 85). The most severe scenario based on SSP585 and the longer timeframe produced declines in Aridity Index with Relative Severity of 69.7-71.3% across 30% of the extent, just below the VU threshold of RS 80%). *Tasmanian Alpine Sphagnum Bog and Associated Fen* was therefore assessed as **Least Concern** (plausible range Least Concern – Near Threatened) under sub-criterion C2b.

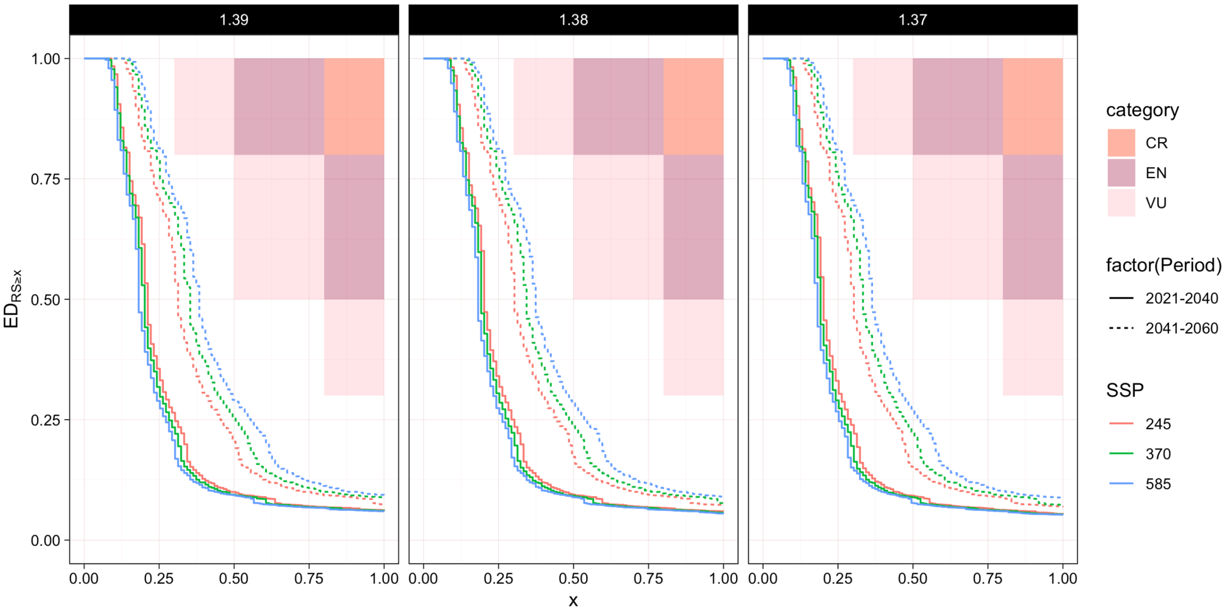


Figure 85. Cumulative degradation function for projected future trends in climatic moisture balance (Aridity Index, AI) for *Tasmanian Alpine Sphagnum Bog and Associated Fen* with for three plausible collapse thresholds (panels) for three climate change scenarios (SSPs) and two timeframes 1970-2000 to 2021-2040 and 2041-2060 (periods), respectively.

Australia

All the scenarios analysed resulted in estimated declines in moisture balance that align with Least Concern status (Figure 86). The most severe scenario based on SSP585 and the longer timeframe produced declines in Aridity Index with Relative Severity of 25-27% across 80% of the extent, just below the VU threshold of RS 80%). *Australian mainland Alpine Sphagnum Bog and Associated Fen* was therefore assessed as **Least Concern** (plausible range Least Concern – Near Threatened) under sub-criterion C2b.

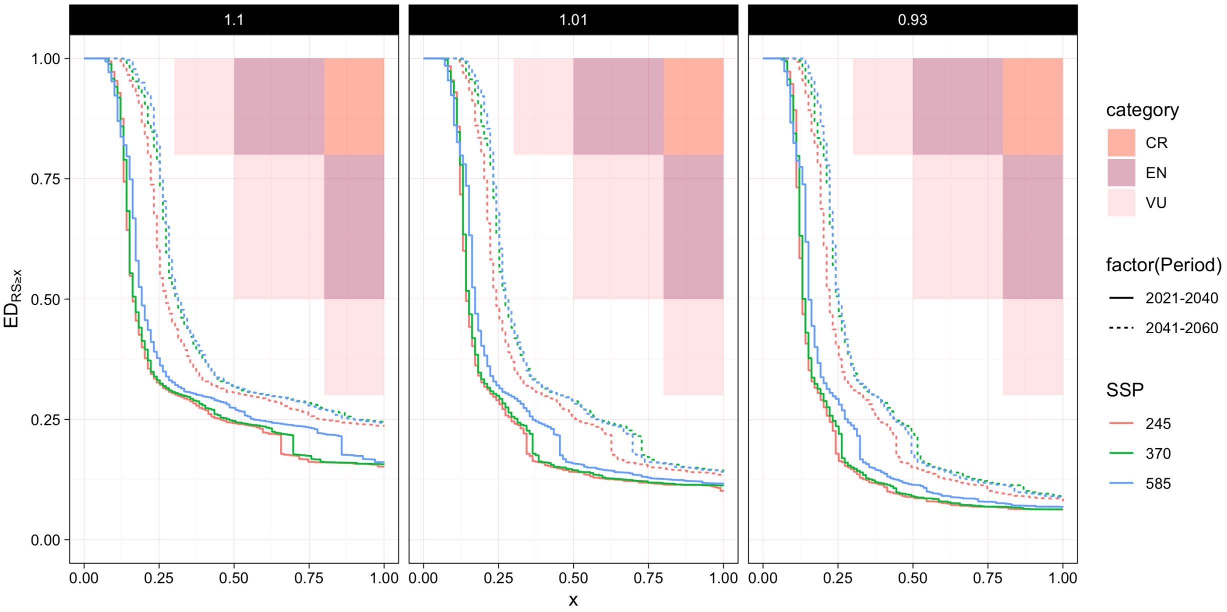


Figure 86. Cumulative degradation function for projected future trends in climatic moisture balance (Aridity Index, AI) for *Tasmanian Alpine Sphagnum Bog and Associated Fen* with three plausible

collapse thresholds (panels) for three climate change scenarios (SSPs) and two timeframes 1970-2000 to 2021-2040 and 2041-2060 (periods), respectively.

Historical change (C3)

Mainland Australia

None of the scenarios analysed resulted in estimated declines in moisture balance that met the thresholds for Vulnerable status (Figure 87). *Mainland Australian Alpine Sphagnum Bog and Associated Fen* was therefore assessed as **Least Concern** (plausible range Least Concern – Least Concern) under sub-criterion C3.

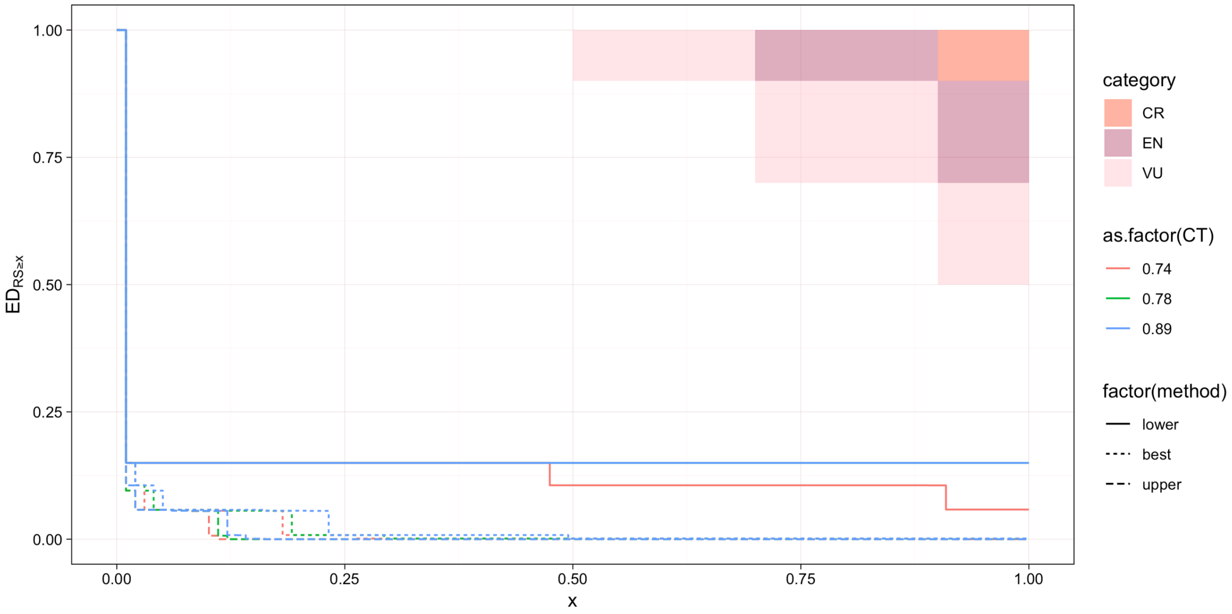


Figure 87. Cumulative degradation function for historic trends in climatic moisture balance (Aridity Index, AI) for *Mainland Australian Alpine Sphagnum Bog and Associated Fen* with the best estimate and 95% lower and upper confidence intervals of the time series regressions for three plausible collapse thresholds (CT 0.74 – 0.89, based on 1st, 2.5th and 5th percentiles of AI across the distribution at the time of mapping). The functions have very large steps due to the large spatial resolution of the raw CRU data.

Tasmania

The best estimate and upper quantile for the most precautionary threshold of collapse intersect the relative severity and extent of degradation thresholds for VU status (Figure 88). *Tasmanian Alpine Sphagnum Bog and Associated Fen* was therefore assessed as **Least Concern** (plausible range Least Concern – Vulnerable) under sub-criterion C3.

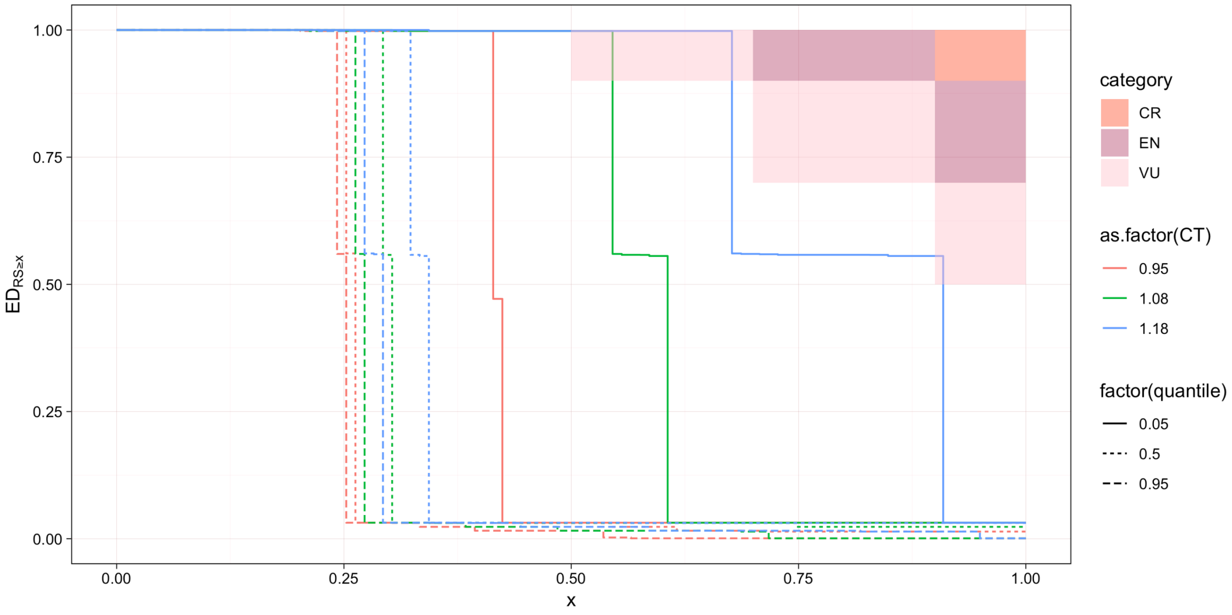


Figure 88. Cumulative degradation function for historic trends in climatic moisture balance (Aridity Index, AI) for *Tasmanian Alpine Sphagnum Bog and Associated Fen* with the best estimate and 95% lower and upper confidence intervals of the time series regressions for three plausible collapse thresholds (CT 1.13 – 1.33, based on 1st, 2.5th and 5th percentiles of AI across the distribution at the time of mapping). The functions have very large steps due to the large spatial resolution of the raw CRU data.

Australia

None of the scenarios analysed resulted in estimated declines in moisture balance met the thresholds for Vulnerable status (Figure 89). *Alpine Sphagnum Bog and Associated Fen in Australia* was therefore assessed as **Least Concern** (plausible range Least Concern – Least Concern) under sub-criterion C3.

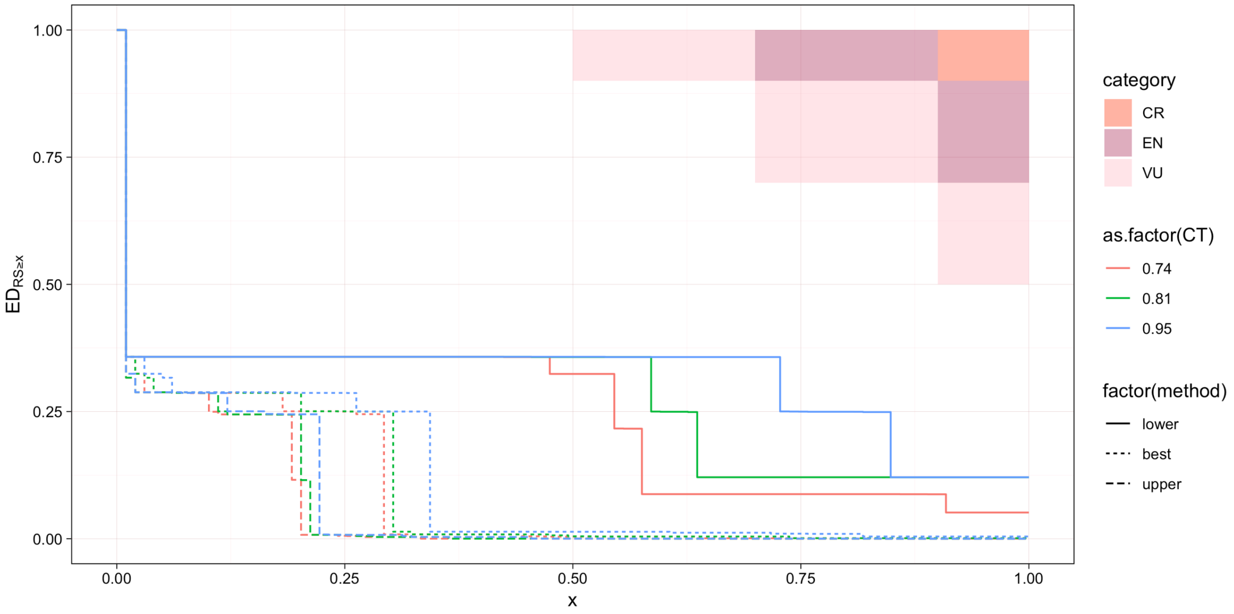


Figure 89. Cumulative degradation function for historic trends in climatic moisture balance (Aridity Index, AI) for *Alpine Sphagnum Bog and Associated Fen in Australia* with the best estimate and 95% lower and upper confidence intervals of the time series regressions for three plausible collapse thresholds (CT 0.74 – 0.95, based on 1st, 2.5th and 5th percentiles of AI across the distribution at the time of mapping). The functions have very large steps due to the large spatial resolution of the raw CRU data.

Indicator: Fire and drought co-occurrence

Relevance to ecosystem function

Both drought and fire pose a risk to *Alpine Sphagnum Bog and Associated Fen*, but when they coincide may result in long-lasting ecosystem degradation or collapse. When prolonged droughts precede fires, there is an elevated risk of peat combustion and/or Sphagnum mortality (Prior et al. 2024). Post-fire drought is also likely to limit regeneration, eroding the compositional diversity and structural integrity of the ecosystem. Although it is not possible to model these processes directly at this time, here, we undertook a simplified pilot analysis to explore the feasibility of indicators and potential risks associated with this threat. Suitable data for the analysis were available only for mainland bogs. We assumed that co-occurrence of fire and drought in the same year would produce insights into both mechanisms of threat. We used climate projections and fire simulations from a run of 20 years to estimate the probability of fire and drought co-occurrence at present and 50 years into the future

Selection of collapse threshold

We assumed one fire within the same season as a severe drought is projected may cause ecosystem collapse based on observations of Wahren et al. (1999), Good (2006), Keith et al. (2023), and Prior et al. (2024).

Data availability and quality

Recent change (C1) and Historical change (C3)

This indicator was not assessed under recent and historical periods.

Future (C2a)

We obtained simulations of future fires for mainland Australian alpine and subalpine ecosystems for 2060 to 2079 from the landscape fire modelling framework 'Fire Regime and Operations Tool' (FROST), detailed in the main methods.

We also obtained climate projections from 1990-2009 at a 10 km resolution from the 'NARCliM' project (Finkele et al. 2006; Evans et al. 2014). We used spatial layers of Daily Drought Factor from 1990-2009, from four Regional Climate Models: the ECHAM5 Regional Climate Model (RCM) 1 (ECHAM-R1), ECHAM5 RCM2 (ECHAM-R2), CSIRO Mk3 RCM1 (CSIRO-R1), and CSIRO MK3 RCM3 (CSIRO-R3). Drought Factor is an index from 0-10 calculated based on soil moisture deficit (the amount of water needed to replenish soil moisture to capacity) and the past 20 days rainfall (Finkele et al. 2006). We determined a relative drought value for the *Alpine Sphagnum Bog and Associated Fen* by obtaining the maximum drought factor value from the daily values across the 20-year period for each cell within the ecosystem type, and subsequently the smallest maximum value from across these cells. This produced a drought factor value of 8, which we assumed as a threshold indicating a severe impact on bogs.

We then used the equivalent NARCLiM spatial layers of Daily Drought Factor to those assessed for recent change, but projected to the time period of 2060-2079. These included four Regional Climate Models: the ECHAM5 Regional Climate Model (RCM) 1 (ECHAM-R1), ECHAM5 RCM2 (ECHAM-R2), CSIRO Mk3 RCM1 (CSIRO-R1), and CSIRO MK3 RCM3 (CSIRO-R3).

Selection of initial and future values

Future change (C2)

We estimated the chance of co-occurrence of fire and drought by assessing the probability of yearly fire and drought occurrence within a 20-year period averaged across 100 simulations. For the period of 2060-2079, we obtained the frequency of fires predicted for the current distribution of *Alpine Sphagnum Bog and Associated Fen*. We extracted a spatial raster that displayed area burnt once within the 20-year period, for each 180 m^2 cell. We then obtained the frequency of years in which drought was predicted across the 20-year period, from 0-20 times. We matched each burnt cell with projected drought frequency to produce a spatially explicit layer of likelihood of the co-occurrence of fire and drought: where 20/20 indicated drought would occur in every year and thus fire and drought were predicted to co-occur (i.e. relative severity 100%), and 10/20 indicated a 50% chance that drought and fire would co-occur (a relative severity of 50%) , assuming a linear relationship between frequency and relative severity of degradation.

Calculation of severity and extent

Future Change (C2)

The extent of projected fires between the four climate scenarios for 2060-2079, ranging from 2.69% to 16.29%% of the ecosystem extent (Table 36). Drought was expected in most years in the areas that burnt, under all scenarios (8-20; Table 2; Column 2).

Under the scenario ECHAM R2, 15.79% of *Alpine Sphagnum Bog and Associated Fen* exceeded the collapse threshold (i.e. relative severity = 100%), experiencing a co-occurrence of drought and fire (20/20; Table 36). This value was 0 % under CISRO R1, 2.32 % under CSIRO R3, and 13.95% under ECHAM R1. Under CSIRO 3, ECHAM R2 and ECHAM R1, all the burnt area (14.26-16.29% of the mainland extent) was projected to experience drought between 15 and 20 years (Relative severity 75-95%; Table 36).

Under these three climate scenarios 13-16.29% of the mainland distribution experienced a relative severity of 80% of more (16-20). This is considered representative of the whole distribution of the ecosystem type. Therefore, this ecosystem is **Least Concern** under sub-criterion C2.

Table 36. Percentage of *Alpine Sphagnum Bog and Associated Fen on mainland Australia* that is projected to burn, and Percentage of Alpine-subalpine Fen that is projected to both burn and experience drought (in how many years) for each climate scenario.

Climate scenario	**Area burnt (%) in 20-year period (Mean across 100 replicates)**	**Number of years (0-20) that drought is projected**	**Area burnt (%, column 1) affected by drought in specified years**	**Mainland extent (%) affected by drought and fire in specified years**
CSIRO R1	2.69%	8	2.32%	0.06%
		10	0.66%	0.02%
		11	2.16%	0.06%
		12	0.33%	0.01%
		13	7.79%	0.21%
		14	0.17%	0.005
		16	5.97%	0.16%
		17	70.98%	1.91%
		18	8.13%	0.22%
		19	1.49%	0.04%
CSIRO R3	14.26%	15	8.83%	1.26%
		16	1.02%	0.15%
		17	3.67%	0.52%
		18	43.56%	6.21%
		19	26.64%	3.80%
		20	16.27%	2.32%
ECHAM R1	14.45%	19	3.46%	0.50%
		20	96.54%	13.95%
ECHAM2	16.29%	18	0.16%	0.03%
		19	2.93%	0.48%
		20	96.91%	15.79%

Indicator: Number of dry months

Relevance to ecosystem function

Alpine Sphagnum Bog and Associated Fen occurs in permanently wet areas (Costin et al. 2000; McDougall & Walsh 2007; Lawrence et al. 2009). When the ecosystem type becomes drier, such as due to draining for construction, droughts, increases the risk of the ecosystem type burning (Prior et al. 2020) and desiccation of *Sphagnum* (Wimbush 1970) and thus local collapse. The higher temperatures and altered precipitation regimes predicted under climate change are likely to alter the moisture levels and threaten the ecosystem type's persistence (Hughes 2003). Here, the number of dry months was used as an indicator of available moisture and susceptibility to burning.

Data availability and quality

Recent change (C1) and Historical change (C3)

There are insufficient data to estimate recent or historical change in the number of dry months.

Future change (C2b)

We used time series of remote sensing products to describe trends in rainfall and evapotranspiration for the period between 2000 and the present. We combined the Climate Hazards Group InfraRed Precipitation with Station data archive (CHIRPS, v2.0, monthly values from 1981 to the present, spatial resolution of 0.05 degrees), and time series of the Evapotranspiration product from the Collection 6 Terra and Aqua Moderate Resolution Imaging Spectroradiometer (MODIS; Product MOD16A2, 8-day composites, 500m spatial resolution). The CHIRPS dataset combines readings from remote sensors with climatic station data to estimate precipitation, and the MOD16A2 applies a modified Penman-Monteith equation to meteorological reanalysis data and remotely sensed vegetation properties. Thus, these time series provide high resolution estimates of changes in environmental conditions (both climate and vegetation) at the sites of interest. Original data were reprojected, low quality measurements were checked, invalid/missing data filled using spatial and temporal interpolation, and time series were then harmonized to the same temporal and spatial resolution. We calculated the number of dry months (months where precipitation < evapotranspiration) of the whole time series for each cell with known occurrence of the ecosystem type. For a subset of cells, we compiled field observations on general bog condition (64 "poor", 66 "medium" and 164 "good") and whether bogs were recently affected by fire or not (319 burnt and 50 unburnt) (Arn Tolsma, pers. comm.; McDougall 2007; Clarke et al. 2015). We examined the relationship between number of dry months and elevation, field condition and burnt status using a linear regression model with interactions between terms.

Selection of collapse threshold

Due to the lack of significant correlation with field measurements of bog condition or fire history, we could not determine a meaningful collapse threshold.

Calculation of initial and present/future values

Future change (C2b)

The number of dry months was calculated for the current conditions (2000 and 2019). This period includes years of very low rainfall (2002 – 2009) followed by high to very high rainfall (2010 – 2016). Precipitation in previous years (1981-1999) were mostly between these extremes. Thus, this value could be used for calculation of sub-criterion C2b under the assumption that these are representative of a larger timeframe including past, present, and future. Time series of remote sensing data show decadal fluctuations, which might confound long term trends in climate with more complex multi-annual cycles. For the analysis we assume a linear relationship with bog condition, but this was not supported by the data.

Calculation of severity and extent

Future change (C2b)

The model was significant (F=77.47, df=7 and 286, $p < 0.005$; $R^2_{adj} = 0.646$), but only elevation and the interaction between elevation and burnt status had significant negative effects, while the other variables and first and second order interactions had non-significant effects. Therefore, there was no clear relationship between the calculated values of number of dry months from remote sensors and the field observations of bog condition or fire history. We were thus not able to propose a meaningful value for the collapse threshold and make the calculations of relative severity and extent. The assessment outcome was therefore **Data Deficient** for sub-criterion C2b. Future assessment should consider alternative datasets and more extensive field monitoring data to provide better descriptions of changes in bog condition through time.

Indicator: Fire frequency

Relevance to ecosystem function

Fires are recognised as one of the principal threats to *Alpine Sphagnum Bog and Associated Fen*. Low intensity fires consume the surface layer of vegetation but have a limited impact on surface *Sphagnum* and peat, and exposed bogs show signs of recovery in composition and *Sphagnum* cover after 3-10 years of the fire event. In contrast, high intensity fires can burn deeply and consume accumulated peat, leading to directional change in composition and structure (McDougall 2007; Hope & Nanson 2015; Clarke et al. 2015).

Selection of collapse threshold

Due to the lack of significant correlation between *fire frequency* and field observations of the effects of fire on bog condition, we could not determine a collapse threshold for this indicator. To assess this indicator, a clear understanding is needed of the relationship between the occurrence of fires and the proportion of *Sphagnum* killed by the fire.

Data availability and quality

Recent change (C1)

We analysed a time series of remote sensing products to estimate the frequency of fires for the period between 2000 and the present. We queried the LANCE Fire Information for Resource Management System (FIRMS) which provides information on active fires detected by the MODIS sensors (Giglio et al. 2003). We reprojected the point location of detected fires, excluded observations with low confidence and used a kernel estimator of density of the spatial point pattern as an estimate of fire frequency per location. Although the correlation between this estimate and the field observations on the incidence of fires was significant and positive (Pearson's r=0.284, t=6.984, df=554, $p < 0.001$, n=556), there were no positive significant correlations with the field observations on the % of bog burnt (n=406), % of *Sphagnum* burnt (n=401) and minimum burnt *Sphagnum* as percentage of bog area (n=138) in Victoria (data from Arn Tolsma, pers comm.). Therefore, the ecosystem type is **Data Deficient** for the indicator of *fire frequency*.

Future change (C2)

Initial values of fire intensity were calculated directly from the time series of FIRMS data for the period 2001 to 2019. Predictions of future changes in fire risk and frequency of high intensity fire are available for selected climatic stations (Lucas et al. 2007) that could be used to estimate probable rates of increase in fire intensity for the year 2050. However, as a reliable collapse threshold could not be set, the ecosystem type is **Data Deficient** under this sub-criterion.

Historical change (C3)

There are insufficient data to reliably estimate historical change in fire frequency, and thus the ecosystem type **Data Deficient.**

Criterion D: Disruption of biotic processes and/or interactions

Summary

Alpine Sphagnum Bog and Associated Fen on mainland Australia were assessed as **Least Concern** (plausible bounds Least Concern – Vulnerable) under criterion D1. Of three biotic indicators assessed, only the prevalence of shrubs showed appreciable declines over the past 50 years. *Alpine Sphagnum Bog and Associated Fen in Tasmania* and *Alpine Sphagnum Bog and Associated Fen in Australia* were **Data Deficient** for criterion D.

Identification of biotic indicator

Vegetation of *Alpine Sphagnum Bog and Associated Fen* is characterised by Sphagnum moss, shrubs and restiads (cordrushes), as well as a range of other herbaceous plant taxa. Ecosystem degradation may result in loss of any of these major biotic components. We therefore examined trends in the prevalence of Sphagnum moss, shrubs and restiads as biotic indicators of collapse.

Relevance to ecosystem function

Sphagnum is an ecosystem engineer that reduces the growth of other plant species and creates acidic conditions that promote its own growth (van Breemen 1995; Hope et al. 2012). *Sphagnum* is the primary peat-forming organism in *Alpine Sphagnum Bog and Associated Fen* and therefore its presence and abundance are a direct measure of ecosystem health, structural integrity, hydrological processes, and function as a carbon sink. Restiads or 'cordrushes' (family Restionaceae) also contribute to peat accumulation and are characteristic of the ecosystem type. Sclerophyll shrubs, primarily of families Ericaceae and Myrtaceae, are also an important structural and functional component of alpine bog vegetation, providing food resources for a range of invertebrate fauna, as well as mycorrhizal associations.

Data availability and quality

Recent change (D1)

Prevalence of plant species was monitored in 10 bogs along an altitudinal gradient (1500 – 2000 m above sea level) in Kosciuszko National Park, NSW, in 1960, 1991, 2005, 2007, 2013, 2018 and 2023 (Figure 90) (Clarke et al 2015). The 10 bogs were selected to represent bogs that had avoided the heaviest degradation from cattle grazing that ceased in the areas c. 1960. Species' prevalence was measured as frequency of occurrence within 25 randomly located replicate quadrats per bog at each census date. Quadrat dimensions were 25 x 10 cm. For this analysis, we assessed three plant functional groups that were represented by species that are characteristic of the alpine bogs and commonly represented across the sites: 1) hummock-forming moss represented by *Sphagnum cristatum*, shrubs represented by *Baeckea gunnii, Epacris glacialis, E. paludosa* and *Richea continentis*; and 3) restiads represented by *Empodisma minus* and *Baloskion australe*. We extracted presence and absence records from the quadrat data for these species and aggregated them into each of these groups for each site and each census date. We assumed that the time series data from Kosciuszko National Park was representative of trends in these plant indicators across the distribution of alpine bogs on mainland Australia, but not necessarily Tasmania, which has a somewhat different history of fire, feral animal activity and grazing. Therefore, the indicator data were applied only to the assessment of Sub-criterion D1 for *Alpine Sphagnum Bog and Associated Fen of mainland Australia*.

Future change (D2) and Historical change (D3)

No data were available to estimate future or historical change in biotic properties of alpine bogs. Therefor sub-criteria D2 and D3 were **Data Deficient**.

Selection of collapse threshold

Alpine Sphagnum Bog and Associated Fen is considered collapsed if any of the three major plant functional groups fall to trace levels defined as 5% prevalence (i.e. present within less than 5% of quadrats censused). We examined sensitivity of assessment outcomes to this threshold of collapse by examining bounds defined by 0% and 10% prevalence, respectively.

Calculation of initial and present/future values

Recent change (D1)

We estimated prevalence for each of the three indicators groups initially by fitting mixed linear models with a binomial error distribution and logit link function to the time series (1964-2023) of quadrat-level presence/absence data, with year as the independent variable and bog site as a random factor. After checking model diagnostics, we identified a need to rescale the fixed predictor (year), and address overdispersion of the data. We rescaled the initial year to 1975, and refitted the model with a negative binomial distribution. This resolved identifiability and reduced correlation in the random effects, but the negative binomial was a poorer fit, so we retained the binomial error distribution with the other adjustments. We used this model to predict the proportion of quadrats occupied for years 1975 and 2025. We examined the sensitivity of assessment outcomes to uncertainty in the trends by using the mean estimate and upper and lower confidence interval in the calculation of relative severity of degradation for each of the three indicators. We also examined variation in trends among the 10 bog sites sampled.

Calculation of severity and extent

Recent change (D1)

The overall trend in *Sphagnum* prevalence across all bogs was marginally positive, but not different from zero (model coefficient for year = 0.0082±0.0091, P=0.31). *Sphagnum* prevalence increased or remained stable in eight of the ten bogs monitored. In the remaining three bogs *Sphagnum* prevalence declined by 55% in two and 10% in one, respectively. Irrespective of whether collapse thresholds were set at 0, 5% or 10% *Sphagnum* prevalence, relative severity of degradation was zero or marginally positive in eight bogs, and greater than 30% in only 20% of bogs sampled. Averaged across all bogs and weighted by quadrats surveyed, *Sphagnum* degradation had a relative severity of 0.7(0-2.2)% for 100% extent. This combination of severity and extent of decline aligns with **Least Concern** status for *Alpine Sphagnum Bog and Associated Fen of mainland Australia* under Sub-criterion D1 based on the best estimate and upper and lower bounds of the estimates. The status of *Alpine Sphagnum Bog and Associated Fen in Tasmania* and *Alpine Sphagnum Bog and Associated Fen in Australia* is **Data Deficient**.

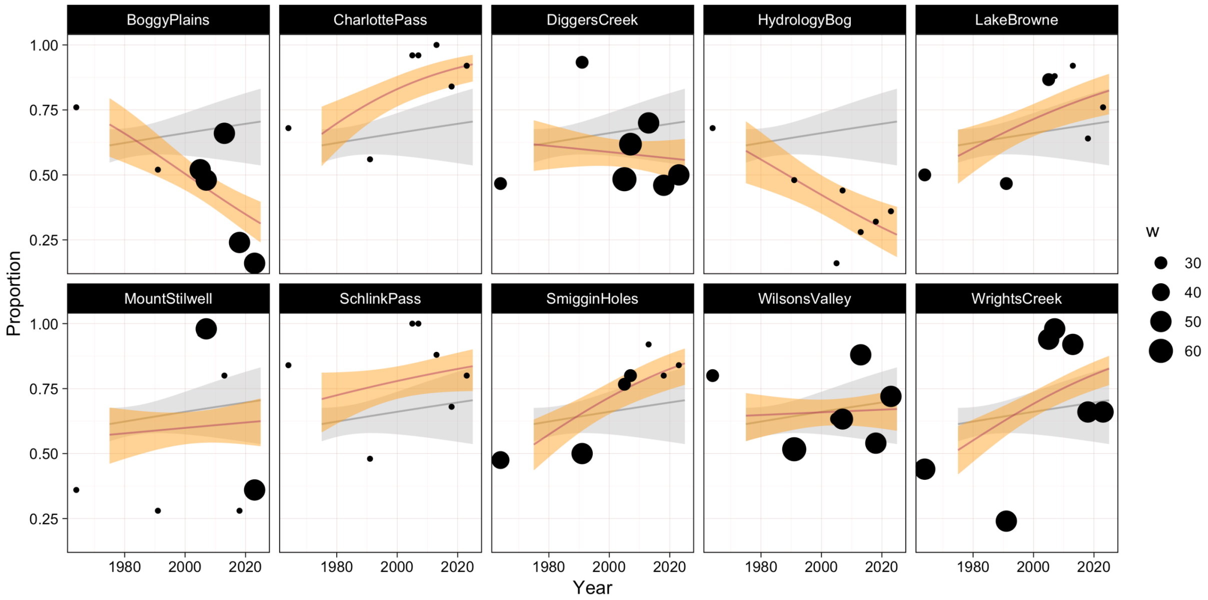


Figure 90. Modelled trends in Sphagnum prevalence (frequency of occurrence in quadrats) with the fixed effect trend (grey ribbon, same for all sites) and the conditional predictions for each bog site based on the combination of fixed and random effects (orange ribbon).

The overall trend in shrub prevalence across all bogs was negative (Figure 91). Shrub prevalence declined moderately or marginally in six of the ten bogs monitored. Irrespective of whether collapse thresholds were set at 0, 5% or 10% Shrub prevalence, relative severity of degradation was greater than 30% in 20-30% of bogs sampled and greater than 50% in 10% of bogs sampled. Averaged across all bogs and weighted by quadrats surveyed, *Sphagnum* degradation had a relative severity of 18 (7-33)% for 100% extent. This combination of severity and extent of decline aligns with **Least Concern** status (plausible bounds Least Concern – Vulnerable) for *Alpine Sphagnum Bog and Associated Fen of mainland Australia* under Sub-criterion D1 based on the best estimate and upper and lower bounds of the estimates. The status of *Alpine Sphagnum Bog and Associated Fen in Tasmania* and *Alpine Sphagnum Bog and Associated Fen in Australia* is **Data Deficient**.

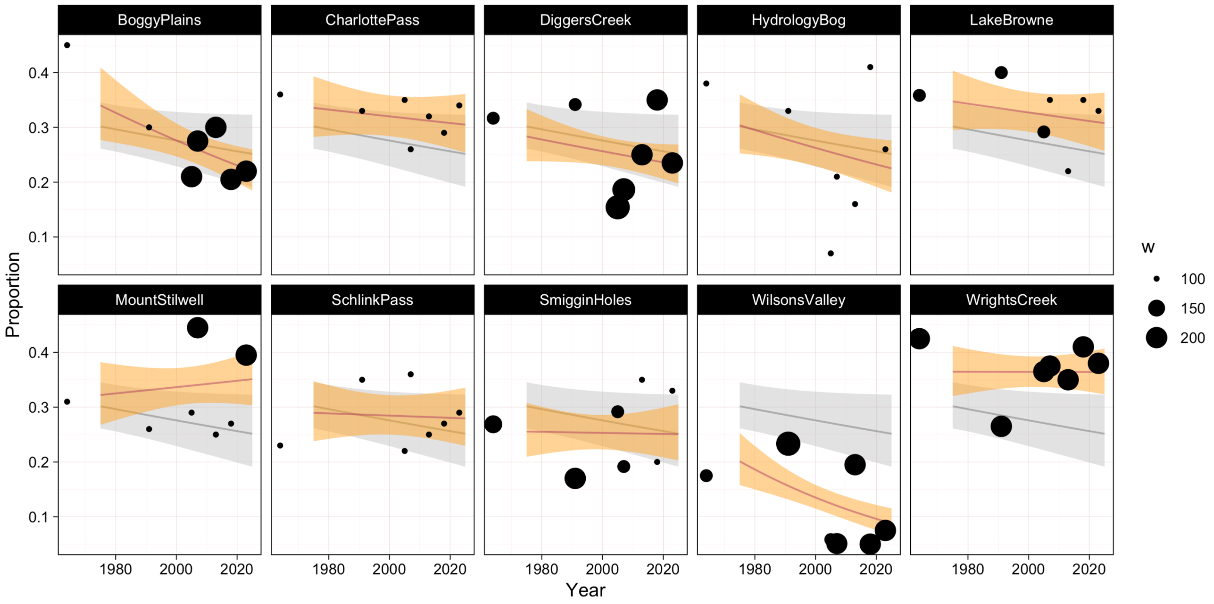


Figure 91. Modelled trends in bog shrub prevalence (frequency of occurrence in quadrats) with the

fixed effect trend (grey ribbon, same for all sites) and the conditional predictions for each bog site based on the combination of fixed and random effects (orange ribbon).

The overall trend in restiad (cordrush) prevalence across all bogs was not different from zero (model coefficient for year = 0.0021±0.0025, P=0.4). Restiad prevalence remained stable or increased in eight of the ten bogs monitored (Figure 92). Irrespective of whether collapse thresholds were set at 0, 5% or 10% restiad prevalence, relative severity of degradation was more than 30% in 20% of bogs. Averaged across all bogs and weighted by quadrats surveyed, *Sphagnum* degradation had a relative severity of 8 (6-10)% for 100% extent. This combination of severity and extent of decline aligns with **Least Concern** status for *Alpine Sphagnum Bog and Associated Fen of mainland Australia* under Sub-criterion D1. The status of *Alpine Sphagnum Bog and Associated Fen in Tasmania* and *Alpine Sphagnum Bog and Associated Fen in Australia* is **Data Deficient**.

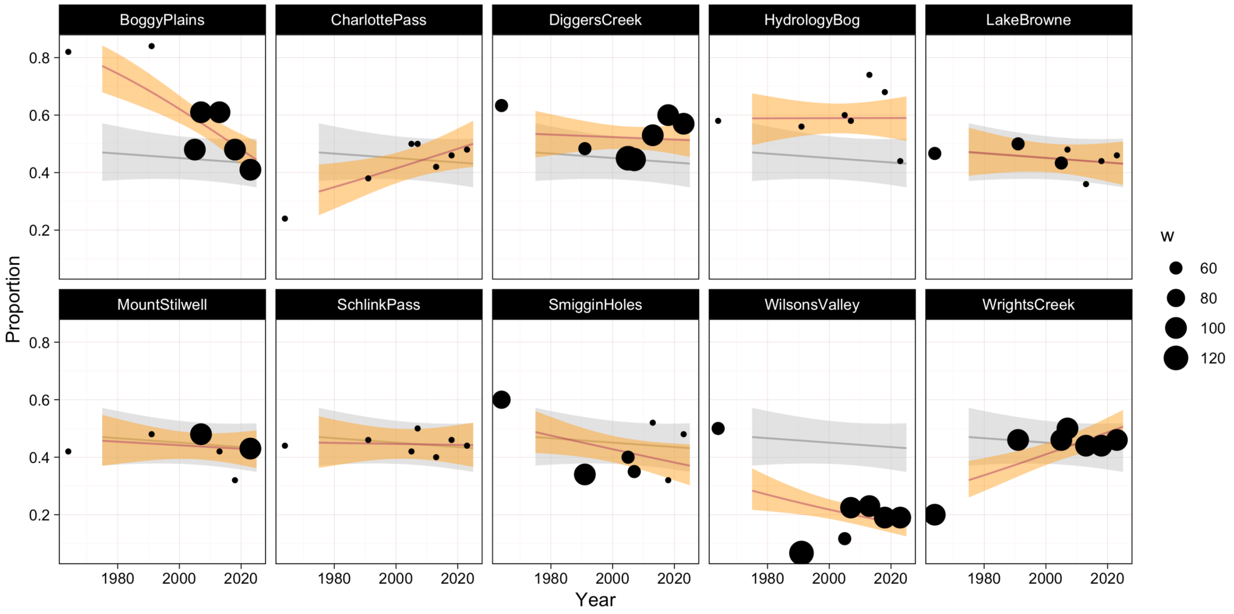


Figure 92. Trends in prevalence of restioids (cordrushes) in ten monitored bogs on Koscuiszko plateau, mainland Australia, 1964 – 2023.

Criterion E: Quantitative analysis of probability of collapse

No stochastic models of *Alpine Sphagnum Bog and Associated Fen* are available and there are insufficient data to quantitatively estimate the probability of collapse within the next 50 to 100 years. Therefore, the risk status is **Data Deficient** under criterion E.

Bog with *Sphagnum* fringing fen on Boggy Plains Creek (Kosciusko National Park). Photo: Keith McDougall.

Alpine-subalpine Fen

Authors

Emily Nicholson, Courtney Taylor, Ella Plumanns Pouton, Susanna Venn, Chloe Sato, Jessica Rowland, Jennie Whinam, Matt White

Reviewers

Arn Tolsma, Courtney Taylor, Anikó B. Tóth

Biome

TF1 Palustrine wetlands

Functional group

TF1.5 Boreal, temperate and montane peat bogs

IUCN status

Data Deficient

Sphagnum hummocks (bog) within *Alpine-subalpine Fen*, Wild Horse Plain, Kosciuszko National Park. Photo: Keith McDougall.

Assessment Summary

Alpine-subalpine Fen occurs in the valley bottoms of the mountainous environments of south-eastern Australia in Tasmania, the Australian Capital Territory (ACT), and New South Wales (NSW). *Alpine-subalpine Fen* is characterised by shallow and permanent to semi-permanent standing water and vegetation dominated by the sedge *Carex gaudichaudiana*, alongside minor amounts of inundation-tolerant herbs, and occasionally *Sphagnum* moss and *Ericaceae* shrubs around the transitional edges. *Alpine-subalpine Fen* occurs in a complex with other water-dependent ecosystems, including *Alpine Sphagnum Bog and Associated Fen*, T*asmanian Alpine Sedgeland*, and *Alpine-subalpine Damp Valley Grassland and Rushland.* The main threatening process to *Alpine-subalpine Fen* is a reduction in groundwater due to climate change, leading to declines in standing water volume and to the cover of the dominant (and inundation-dependent) *Carex gaudichaudiana*. Other threats include feral ungulates, weed invasion, and fire in the surrounding catchment. Most criteria, including those relating to the key threatening processes, are assessed as **Data Deficient,** because many components of this ecosystem type are understudied, making it difficult to predict its response to a changing climate (Table 37). The status of this ecosystem type is assessed as **Data Deficient**, because there was insufficient evidence to assign a risk category, given the lack of data for the most pertinent criteria.

Table 37. Summary of the Red List of Ecosystems assessment of the *Alpine-subalpine Fen*. Category ranges in parentheses reflect uncertainty in assessment under the corresponding criteria.

Criteria	A	B	C	D	E	Overall
Sub-criterion 1 *A, C, D: past 50-years* *B: EOO*	DD	LC	DD	DD	DD	DD
Sub-criterion 2 *A, C, D: 50-year period including present & future* *B: AOO*	DD	LC	LC	DD		
Sub-criterion 3 *since ~1750* *B: number of locations*	DD	LC	DD	DD		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria

Ecosystem Description

Ecosystem Classification

In the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022a), *Alpine-subalpine Fen* belongs to Ecosystem Functional Group *TF1.6 Boreal, temperate and montane peat bogs*, within the *Palustrine Wetlands* biome. *Alpine-subalpine Fen* is classified in multiple classifications across south-eastern Australia. It is referred to as “Fen” (Costin et al. 2000; McDougall & Walsh 2007; Venn et al. 2017), or can be “*Carex* Fen” in NSW (Hope et al. 2012) and ACT (Keith 2004). In Tasmania, the ecosystem type is classified as “Fen” and forms part of the Freshwater Aquatic Sedgeland and Rushland (unit ASF) community (Kirkpatrick 1997; Kitchener & Harris 2013). Fen-related Ecological Vegetation Classes (EVCs) in Victoria (i.e. EVC 171) are included within the ecosystem type *Alpine Sphagnum Bog and Associated Fen* (see mapping methods in main methods).

Distinction from similar ecosystem types

Alpine-subalpine Fen occurs in a complex with other water-dependent ecosystems. This complex has been separated into four different ecosystem types in this assessment: *Alpine-subalpine Fen; Alpine Sphagnum Bog and Associated Fen; Alpine-subalpine Damp Valley Grassland and Rushland;* and *Tasmanian Alpine Sedgeland.* The *Alpine Sphagnum Bog and Associated Fen* assessment was based on the Environment Protection and Biodiversity Conservation (EPBC) listing (Threatened Species Scientific Committee 2009), and thus includes Fen associated with Bog to acknowledge their interdependence. The ecosystem type assessed here, *Alpine-subalpine Fen*, captures standalone Fen, differentiated from *Alpine Sphagnum Bog and Associated Fen* by 1) dominance of *Carex* and similar species (whereas *Alpine Sphagnum Bog and Associated Fen* is dominated by *Sphagnum* moss), and 2) permanent or periodic standing water forming pools (Venn et al. 2017). Two adjacent ecosystem types that are typically drier are: 1) *Alpine-subalpine Damp Valley Grassland and Rushland* (present on the Australian mainland and Tasmania): a continuum of grass- and rush-dominated communities, found in deep fertile soils subject to intermittent seasonal water-logging in valley bottoms (Kirkpatrick 1983); and 2) *Tasmanian Alpine Sedgelands*: a community dominated by hard-leaved monocotyledonous plants that form mats or tussocks.

General international terminology differs substantially from that used in Australia, as the Australian “fen” corresponds most closely with northern hemisphere marsh or mire (Rydin & Jeglum 2015; Venn et al. 2017). The term mire refers to wet systems that are peat forming and includes both Bog and Fen. International literature typically defines bog as a wetland system fed solely by precipitation (ombrogenous) and fen as one which relies on rainwater and groundwater (minerogenous), as this determines their nutrient levels (Rydin & Jeglum 2015). In contrast, the term peatland is used in Australia as a broad term to capture peat-forming communities, which are distinguished based on dominant vegetation type: fen is typified by dominance of graminoids, particularly *Carex* species, and bog by the dominance of *Sphagnum* moss (Hope & Whinam 2005). To align with this approach, Bog and Fen are separated in this assessment by dominant vegetation type and by the characteristic presence of pools of standing water in Fen, rather than by water source.

Distribution

Australian *Alpine-subalpine Fen* occurs in the mountainous environments of south-eastern Australia (ACT, NSW and Tasmania). The ecosystem type is distributed between 145.88° to 148.99° longitude and between -35.50° and -42.22° latitude. The altitudinal band occupied differs between states and territories in Australia: it is found above 1000 m elevation on the mainland (NSW, ACT), and above 800 m elevation in Tasmania. These elevation thresholds are consistent with the definition for *EPBC*-listed *Alpine Sphagnum Bog and Associated Fen*. Due to the special physiographic and nutrient requirements of Fen, it is best developed in the Snowy Mountains and in similar areas of Fen in Tasmania (Kirkpatrick 1984b; Green & Osborne 2012). Spatial products used to create the present-day distribution map represent the most extensive and/or accurate mapping available within each state for the defined bounds of the *Alpine-subalpine Fen* at the time of assessment (July 2021), as advised by representatives from each government department involved in the assessment process. Fens in Victoria have been included in the *Alpine Sphagnum Bog and Associated Fen* assessment. In total, *Alpine-subalpine Fen* covers an area of approximately 15.81 km^2 (Figure 93). This comprises:

- 7.98 km^2 in Tasmania
- 7.35 km^2 in New South Wales
- 0.47 km^2 in the Australian Capital Territory

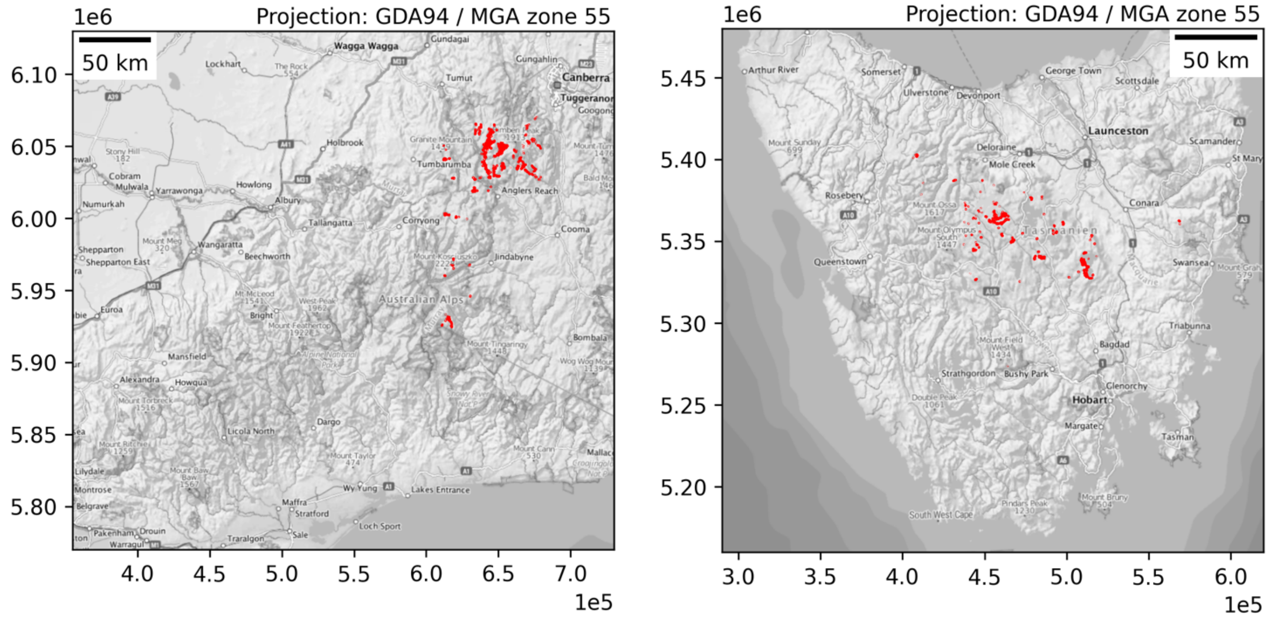


Figure 93. Distribution of the *Alpine-subalpine Fen* (red) across the Australian mainland (left) and Tasmania (right).

Abiotic environment

Alpine-subalpine Fen is generally found in lower parts of the alpine landscape, such as valley basins where cold air accumulates, with a low mean daily minimum temperature in the coldest month of -4.2 °C (Bridle & Kirkpatrick 1999). The topography of these areas also creates poor drainage, allowing surface flows and groundwater to form permanent to semi-permanent pools that vary in size, with some up to 100 m^2 in NSW where this system is best developed (Costin et al. 2000; McDougall & Walsh 2007). Pools are up to 15 cm deep, and remain inundated through most, if not all, of summer, though they may dry out to very damp peaty soils (McDougall & Walsh 2007). *Alpine-subalpine Fen* is underlain by peat, which being highly organic, can accumulate substantial amounts of carbon (for example, a total of 705 tonnes/year in the *Alpine-subalpine Fen* of the Snowy Mountains (Hope et al. 2009). Palaeoecological studies suggest this ecosystem type has persisted throughout the Holocene and contains some of the deepest peatlands in Australia (Hope et al. 2009).

Individual fens may be as large as Sally Tree Creek fen in NSW, which is 250 m wide and follows the valley floor for 2 km, or they can occur as small patches within *Alpine-subalpine Damp Valley*

Grassland and Rushland or as ribbons along streambanks (Hope et al. 2012). *Alpine-subalpine Fen* is acidic to neutral (pH 4.5-6) (Bridle & Kirkpatrick 1999; Hope et al. 2009; Kirkpatrick et al. 2014). The ecosystem type can have a high proportion (~22%) of organic carbon (Kirkpatrick et al. 2014), which in combination with highly saturated soils, limits the species that can grow. *Alpine-subalpine Fen* tends to have high extractable phosphorus (56 ppm) and total phosphorus (1690 ppm) compared with other alpine and subalpine ecosystems (Bridle & Kirkpatrick 1999).

Characteristic native biota

Alpine-subalpine Fen is dominated (> 50% cover) by the vegetatively spreading sedge *Carex gaudichaudiana* (Bridle & Kirkpatrick 1999). Species composition and structure is consistent between mainland Australia and Tasmania (Kirkpatrick 1997). Relatively few other species are found within Fen, but other inundation-tolerant species may be reasonably common in gaps between sedge tussocks or in shallow pools, including *Brachyscome obovata, Carex echinata, Deschampsia caespitosa, Pratia surrepens, Epilobium gunnianum, Isolepis crassiuscula,* and *Myriophyllum pedunculatum* (McDougall & Walsh 2007; Hope et al. 2012). Where *Alpine-subalpine Fen* transitions to other ecosystem types, there may be minor amounts of *Empodisma minus, Poa costiniana,* or shrubs such as *Epacris* or *Dracophyllum* species (Costin et al. 2000).

Fauna are not known to drive major functions in *Alpine-subalpine Fen*, but may be common during summer when the surrounding landscape is dry. Invertebrates such as the *Kosciuscola* grasshoppers are common as they feed on *Carex* leaves (Green & Osborne 2012) and flies are also common. Although invertebrate diversity is poorly studied in Fen, Hope et al. (2012) suggest they may have effects on peatland ecology. Fens provide important breeding grounds for frog species, including the common eastern froglet (*Crinea signifera*) and alpine tree frog (*Litoria verreauxii alpina*). Reptiles are common, such as the alpine water skink (*Eulamprus kosciuskoi*), the mountain swamp skink (*Pseudemoia rawlinsoni*), and the white-lipped snake (*Drysdalia coronoides*) (Steane et al. 2005). Where Fen is connected to streams, fish such as mountain galaxias (*Galaxias olidus*) may be present, particularly at lower altitude (Hope et al. 2012). Mammals are largely absent, but some may move in for feeding in drier conditions. The migratory bird Latham's snipe (*Gallinago hardwickii*) can be found in summer.

Key components, processes, and interactions

Hydrological regimes

Water is the main driver of the distribution and function of *Alpine-subalpine Fen*. This ecosystem type is fed by surface flows and groundwater, creating conditions where the water table is typically at the surface for most of the year. Water from snow melt plays an important role in spring water recharge, after patchy recharge during the rest of the year (Venn et al. 2017). *Alpine-subalpine Fen* plays an important role in flow regime, spreading water through the landscape and trapping sediment, thus regulating water quality (Hope et al. 2012). The flow of groundwater carries minerals into Fen, and together with the standing water and permanent pools, inhibits the growth of vegetation that are intolerant of low nutrient levels and permanent inundation, creating the low diversity vegetation dominated by the sedge *Carex gaudichaudiana*. Standing water allows the formation of fibrous sedge peat, which underlies *Alpine-subalpine Fen* at depths of 2-4 m in some areas (Hope & Nanson 2015) (Figure 94). Moisture retention in *Alpine-subalpine Fen* is essential for continued carbon storage; during the summer, *Alpine-subalpine Fen* is dependent on groundwater inputs for continued moisture. *Alpine-subalpine Fen* provides water and green forage during dryer periods, essential to maintaining faunal biodiversity at the landscape scale. Degraded *Alpine- Sphagnum Bog and Associated Fen* that does not recover from fire damage may transition to *Alpine-subalpine Fens* or *Alpine-subalpine Damp Valley Grassland and Rushland* (McDougall et al. 2023).

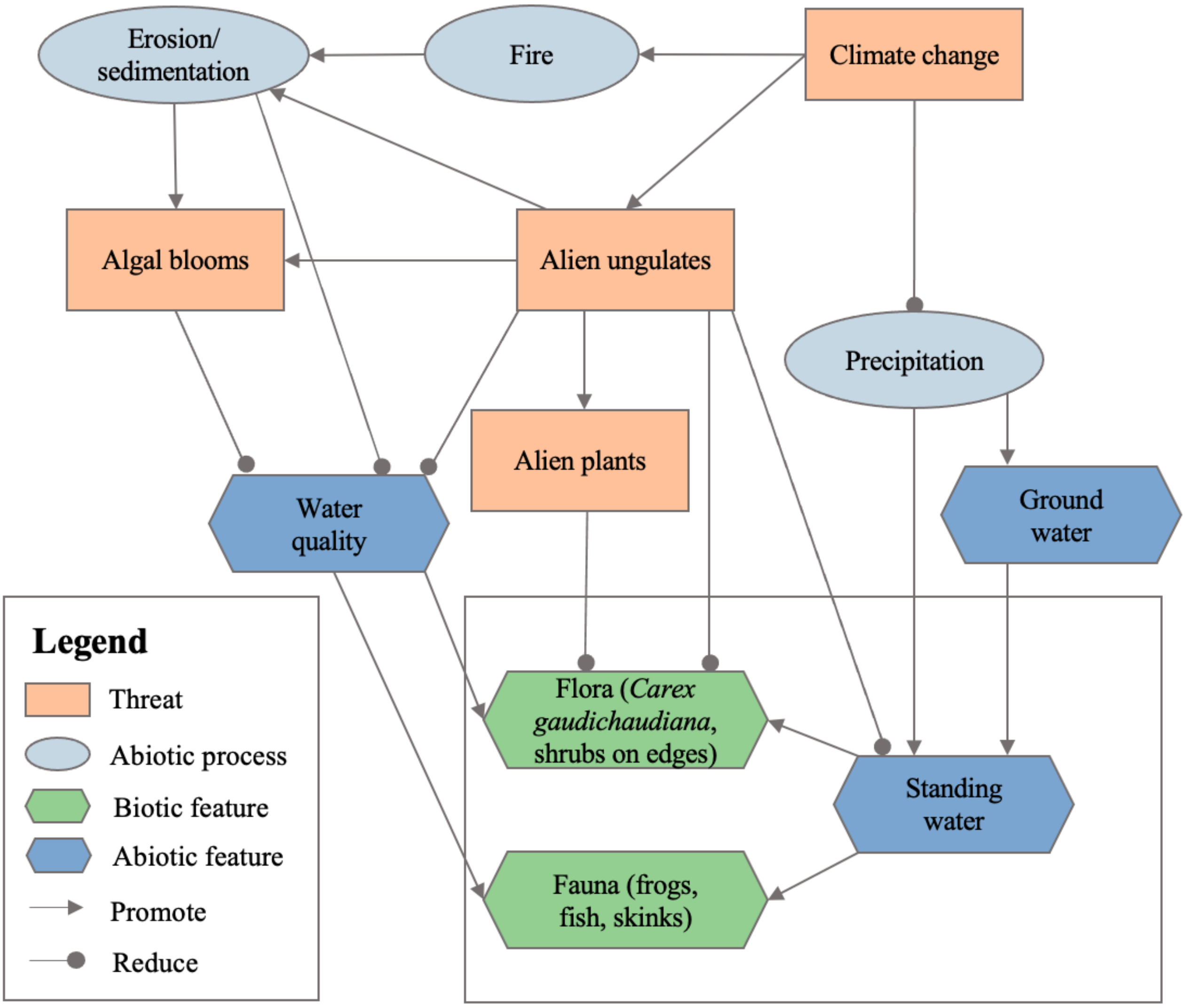


Figure 94. Conceptual diagram illustrating relationships between, and threats to, biotic and abiotic components and processes in *Alpine-subalpine Fen*. Box contains the main biotic and abiotic components.

Major threats

Climate change

Climate change poses one of the greatest threats to the extent and condition of water-dependent ecosystems such as *Alpine-subalpine Fen*. *Alpine-subalpine Fen* is capable of drying and re-wetting over short periods, and subsequently resuming normal ecosystem function. However, the longer-term effects associated with a dryer and warmer climate may threaten this ecosystem type (Worboys et al. 2011). For example, changes to the amount of water inputs from precipitation (including snow melt) and increased evaporation, may reduce water below sustainable levels in the long term, and make them more susceptible to fires. The impact of climate change on Australian alpine soils is not well understood, but increased soil temperatures will likely increase biotic activity and decomposition rates in peat soils (Wilson et al. 2021). Consequently, dryer extents may become a carbon source during extended warm phases, particularly hot summers. However, in *Alpine-subalpine Fen* where the bulk of the peat deposit has already lost readily oxidisable components, they may have some resilience to warming conditions (Hope et al. 2012).

Fire and drought

Fen is relatively resilient to fire as it is a predominantly wet ecosystem type, and the dominant sedge *C. gaudichaudiana* regenerates quickly via re-sprouting after burning (Hope et al. 2009). However, drought in combination with fire may prevent the return to normal ecosystem function. At the landscape scale, a predicted reduction in forest cover due to increased fires may reduce water infiltration and rates of groundwater recharge, leading to increased drying of fen which rely on groundwater in summer (Worboys et al. 2011). It may also increase exposed soil, erosion, and siltation of fens, causing poor water quality which may lead to algal blooms (Mosley 2015).

Alien ungulates

The soft banks of *Alpine-subalpine Fen* are particularly susceptible to damage by hard-hooved ungulates (McDougall & Walsh 2007). Feral horses (*Equus caballus*) and deer (primarily *Rusa unicolor* on the mainland and *Dama dama* in Tasmania) can enter *Alpine-subalpine Fen* to drink and feed on *C. gaudichaudiana*. Grazing and trampling lead to bank incision, channelisation, soil erosion and subsequent sedimentation and siltation issues (Robertson et al. 2019). As *Alpine-subalpine Fen* is located at low points in the landscape, trampling and destruction of edges typically doesn't lead to drainage in the absence of other drivers, but can alter the hydrology. The impacts of alien ungulates may also interact with other threats. For example, during the 2003 wildfires, only the *Alpine-subalpine Fen* that had also experienced drying due to trampling-driven incisions burnt (Hope et al. 2012).

Alien invasive plants

In wet *Alpine-subalpine Fen*, soft rush, *Juncus effusus* can be a high threat weed (McDougall et al. 2005). Dry and drying *Alpine-subalpine Fen* is more susceptible to invasions by both alien and native species (Ashton & Williams 1989). Alien plants may include pasture grasses or *Myosotis laxa subsp. caespitosa* (McDougall & Walsh 2007). Native species less tolerant of inundation such as *Poa costiniana* and *Empodisma minus* can colonise dried out peats, ultimately leading to conversion to *Alpine-subalpine Damp Valley Grassland and Rushland* (Pickering & Armstrong 2003; Grover et al. 2005).

IUCN Stresses Classification

1.2 – Ecosystem degradation
1.3 – Indirect ecosystem effects
2.3 – Indirect Species Effects

IUCN Threats Classification

6.1– Recreational Activities
7.1.1 – Increase in fire frequency / intensity
8.1.2 – Invasive non-native species
11.2 – Droughts
11.3 – Temperature Extremes

Ecosystem collapse

Alpine-subalpine Fen can collapse when it loses standing water for extended periods. The ecosystem type may transition to *Alpine-subalpine Damp Valley Grassland and Rushland* or

Tasmanian Alpine Sedgeland if it is filled with sediment and loses its profile, or when standing water is lost through drying. It also may collapse if weeds invade and become dominant, however this is a theoretical collapsed state and unlikely to occur.

Alpine-subalpine Fen collapses when any of the following occur:

1) *Area*: The mapped extent declines to zero (100% loss) (Criteria A & B), and/or;
2) *Standing water level*: There is a loss of standing water causing peat soils to dry out for extended periods of time (> 3 years) (Criterion C), and/or;
3) *Fire/drought co-occurrence*: Fire co-occurs with drought (pre- or post-fire) across the entire ecosystem extent (Criterion C), and/or;
4) *Sedge cover: Carex gaudichaudiana* cover is < 10 % (Criterion D).

Risk Assessment

Criterion A: Reduction in distribution

Summary

It is unlikely that the extent of *Alpine-subalpine Fen* has decreased by more than 30% in the last 50 years or since European invasion. The extent may have increased in the last 50 years as *Alpine Sphagnum Bog and Associated Fen* that do not recover from fire can transition to this ecosystem type; this may continue in the future. However, there are currently no temporal data to quantify the changes in distribution of this community, due to challenges in mapping the ecosystem type across its distribution. The risk status is **Data Deficient** under sub-criteria A1, A2 and A3.

Criterion B: Restricted distribution

Summary

The risk status of *Alpine-subalpine Fen* is **Least Concern** under all sub-criteria B.

Methods

Under criterion B, the extent of occurrence (EOO; sub-criterion B1) and area of occupancy (AOO; sub-criterion B2) of *Alpine-subalpine Fen* were determined using a combination of existing map products from the Australian Capital Territory (ACT Government 2018), New South Wales (Hope et al. 2012), Victoria (DELWP 2021), and Tasmania (Kitchener & Harris 2013; DPIPWE 2020) – see main methods.

The number of threat-defined locations was based on drought as this is the most important stochastic threat to *Alpine-subalpine Fen* (sub-criterion B3).

Assessment outcome

Sub-criterion B1

The Extent of Occurrence (EOO) of *Alpine-subalpine Fen* is currently estimated at 88,920.45 km^2 (Figure 93). The risk status is **Least Concern** under sub-criterion B1.

Sub-criterion B2

Alpine-subalpine Fen occupies 66 10×10 km grid cells (Area of Occupancy, AOO; Figure 95). The risk status of the ecosystem is **Least Concern** under sub-criterion B2.

Sub-criterion B3

The most serious plausible threat to *Alpine-subalpine Fen* is prolonged periods of drought. However, this is unlikely to cause collapse within a short period of time (c. 20 years) because drought is likely to occur unevenly across the distribution. Additionally, while snow cover and winter rainfall are predicted to decrease, summer rainfall may increase, making it difficult to predict the consequences of climate change in the near future for this ecosystem type. The status of the ecosystem type under sub-criterion B3 is therefore **Least Concern**.

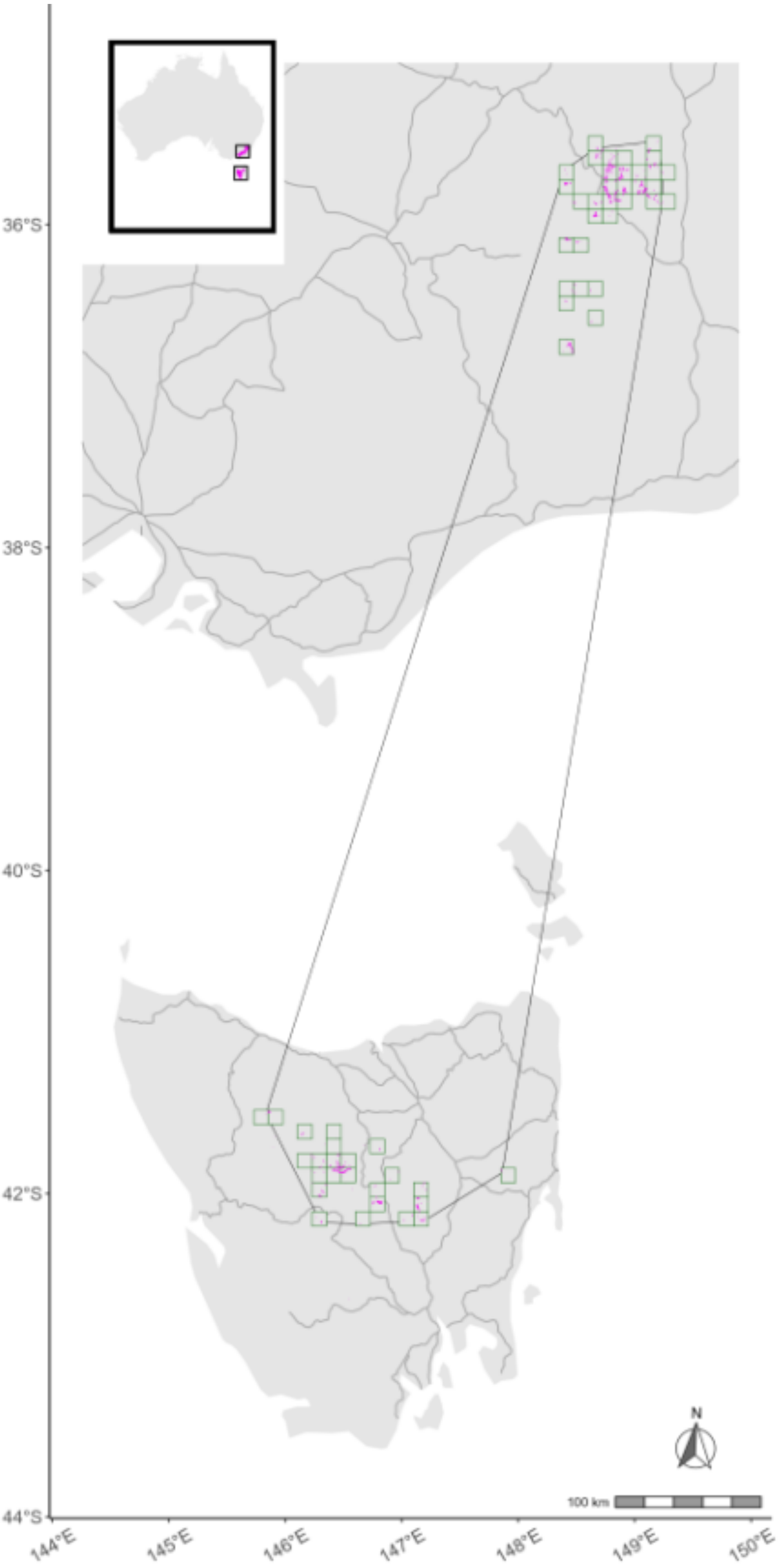


Figure 95. Map of Australian *Alpine-subalpine Fen* (magenta polygons), showing EOO (black polygon) and AOO where the 1% rule was not applied (green squares).

Criterion C: Environmental degradation

Summary

The status of *Alpine-subalpine Fen* is assessed as **Least Concern** under sub-criterion C2, and **Data Deficient** under sub-criteria C1 and C3.

Identification of abiotic indicator

We examined the data availability for two abiotic indicators to assess the risk of collapse from environmental degradation:

- *Standing water level:* a measure of the standing pools of water, characteristic of the ecosystem type.
- *Fire and drought co-occurrence*: A measure of the co-occurrence of fire with drought, capable of killing characteristic vegetation.

Indicator: Standing water level

Relevance to ecosystem function

Alpine-subalpine Fen is characterised by standing pools of water that support the dominant inundation tolerant sedge *Carex gaudichaudiana.* While these pools can dry out for short periods over summer, the soil typically remains very moist, and water is recharged annually. Once standing water is lost this ecosystem type could transition to a drier ecosystem type, most likely *Alpine-subalpine Damp Valley Grassland and Rushland.*

Data availability and quality

No data currently exist to measure changes through time, predict collapse thresholds (i.e. when the level of standing water is too dry and for how long that would result in collapse of this system). Therefore, this ecosystem type is **Data Deficient** under this indicator.

Indicator: Fire and drought co-occurrence

Relevance to ecosystem function

Overly frequent drought and fire both pose a risk to *Alpine-subalpine Fen*, through the reduction of the quantity and quality of available standing water. If fire or drought occur alone at appropriate lengths and intervals, the characteristic vegetation, such as dominant sedge *C. gaudichaudiana* is capable of regeneration (Hope et al. 2009). However, the co-occurrence of fire and drought within the same season is likely to prevent regeneration and thus cause transition to a novel assemblage of plants, or collapse into *Alpine-subalpine Damp Valley Grassland and Rushland.*

Data availability and quality

Recent change (C1) and Historical change (C3)

This indicator was not assessed under recent and historical periods.

Future change (C2a)

We obtained future fire simulations for mainland Australian alpine and subalpine ecosystems for 2060 to 2079 from the landscape fire modelling framework 'Fire Regime and Operations Tool' (FROST), detailed in the main methods.

We also obtained climate projections from 1990-2009 at a 10 km resolution from the 'NARCliM' project (Finkele et al. 2006; Evans et al. 2014). We used spatial layers of Daily Drought Factor from 1990-2009, from four Regional Climate Models: the ECHAM5 Regional Climate Model (RCM) 1 (ECHAM-R1), ECHAM5 RCM2 (ECHAM-R2), CSIRO Mk3 RCM1 (CSIRO-R1), and CSIRO MK3 RCM3 (CSIRO-R3). Drought Factor is an index from 0-10 calculated based on soil moisture deficit (the amount of water needed to replenish soil moisture to capacity) and the past 20 days rainfall (Finkele et al. 2006). The equations are available in Finkele et al. (2006). We determined a relative drought value for the *Alpine-subalpine Fen* by obtaining the maximum drought factor value from the daily values across the 20-year period for each cell within *Alpine-subalpine Fen*, and subsequently the minimum maximum value from across these cells. This produced a drought threshold of a drought factor value of 8.

We then used the equivalent NARCLiM spatial layers of Daily Drought Factor to those assessed for recent change, but projected to the time period of 2060-2079. These included four Regional Climate Models: the ECHAM5 Regional Climate Model (RCM) 1 (ECHAM-R1), ECHAM5 RCM2 (ECHAM-R2), CSIRO Mk3 RCM1 (CSIRO-R1), and CSIRO MK3 RCM3 (CSIRO-R3).

Selection of collapse threshold

As one fire within the same season as a drought is projected to cause ecosystem collapse, we selected one co-occurrence of fire in the year that drought was predicted as the collapse threshold.

Selection of initial and future values

Future change (C2)

We estimated the chance of co-occurrence of fire and drought by assessing the average probability of yearly fire and drought occurrence within a 20-year period. For the period of 2060-2079, we obtained the frequency of fires predicted for the current distribution of *Alpine-subalpine Fen*. We extracted a spatial raster that displayed area burnt once within the 20-year period, for each 180 m^2 cell. This was obtained as an average across 100 simulations. We then obtained the frequency of years in which drought was predicted across the 20-year period, from 0-20 times. We matched each burnt cell with projected drought frequency to produce a spatially explicit layer of likelihood of the co-occurrence of fire and drought: where 20/20 indicated drought would occur in every year and thus fire and drought was predicted to co-occur (i.e. relative severity 100%), and 10/20 indicated a 50% chance that drought and fire would co-occur (a relative severity of 50%) , assuming a linear relationship between frequency and relative severity of degradation.

Calculation of severity and extent

Future change (C2)

Under the four climate scenarios, there were varying spatial extents where fire was predicted to occur between 2060-2079, ranging from 1.54% to 24.98% of the ecosystem extent. Drought was expected in most years in the areas that burnt, under all scenarios (17-20). Under the scenario ECHAM R2, 24.98% of *Alpine-subalpine Fen* exceeded the collapse threshold (i.e. relative severity = 100%), experiencing a co-occurrence of drought and fire (20/20; Table 38 Column 4). This value was 1.53 % under CSIRO R1, 2.53 % under CSIRO R3, and 17.22% under ECHAM R1. All burnt area was also projected to experience drought between 15 and 19 of the 20 possible years (Relative severity 75-95%; Table 38).

Between 1.54 and 24.98 % of the ecosystem extent experienced a relative severity of 80% or more. Therefore, this ecosystem type is **Least Concern** under sub-criterion C2.

Table 38. Percentage of *Alpine-subalpine Fen* that is projected to burn, and Percentage of *Alpine-subalpine Fen* that is projected to both burn and experience drought (in how many years) for each climate scenario.

Climate scenario	**Percentage of ecosystem projected to burn in 20-year period (Mean across 100 replicates)**	**Number of years (0-20) that drought is projected**	**Percentage of the burnt area (column 1) affected by drought in specified years**	**Percentage (%) of entire mainland ecosystem extent affected by drought and fire in specified years**
CSIRO R1	1.54%	17	100%	1.54%
CSIRO R3	15.00%	18	30.28%	4.54%
		19	52.82%	7.92%
		20	16.90%	2.53%
ECHAM R1	17.22%	20	100%	17.22%
ECHAM R2	24.98%	20	100%	24.98%

Criterion D: Disruption of biotic processes and/or interactions

Summary

Directional change in plant species composition and invasive ungulate species (identity and abundance) are the most relevant indicators of biotic disruption in *Alpine-subalpine Fen.* The available data are insufficient to draw a conclusion about rates of biotic disruption. Hence the status of the ecosystem under criterion D is **Data Deficient.**

Identification of biotic indicators

We examined the relevance and data availability for one biotic indicator to assess the risk of collapse from disruption of biotic processes and/or interactions.

- *Plant species composition*: a measure of the relative dominance of characteristic plant species.

Indicator: Plant species composition

Alpine-subalpine Fen is characterised by high cover of the dominant monocot species *Carex gaudichaudiana*. This inundation-tolerant species typically has >50 % cover in this ecosystem type. Climate change may lead to changes in the water balance of this ecosystem type and hence reductions in this dominant species, yet there are currently no time series data to show that this is occurring nor with capacity to assess this indicator. Another driver of change in plant species composition could be invasion by exotic species, but there is currently no evidence of this happening at large scales.

Criterion E: Quantitative analysis of probability of collapse

Assessment outcome

No stochastic models are available and there are presently insufficient data to quantitatively estimate the probability of collapse of *Alpine-subalpine Fen* within the next 50 to 100 years. Therefore, the risk status is **Data Deficient** under criterion E.

Boggy Plains Creek (KNP) showing mosaic on valley floor of damp valley grassland and fen. Photo: Keith McDougall.

Freshwater biome

Photo: …

Alpine-subalpine Streams

Authors

Rebecca E Lester, Michael Traurig, Galen Holt, Emily Armstrong, Jessica Rowland

Reviewers

Emily Nicholson, Tracey Regan, Anikó B. Tóth

Contributors

Arn Tolsma, Ken Green, Jennie Whinam, Emily Nicholson

Biome

F1 Rivers and streams

Functional group

F1.1 Permanent upland streams

IUCN status

Endangered

Assessment Summary

Alpine-subalpine Streams is restricted to elevations of ³ 915 m in Tasmania, ³ 1,370 m in Victoria and ³ 1,500 m in New South Wales. It is defined by a distribution typically above the tree line, a permanent water flow, and characteristic biota that have adapted to the harsh cool climate. Threats to the ecosystem include changes to hydrological regimes and temperatures because of infrastructure development, climate change, and invasive species outcompeting endemic species. The ecosystem type was assessed as Least Concern under Criterion A, B and C, and Vulnerable under Criterion D due to changes in microinvertebrate assemblages. The status of the ecosystem is assessed as **Endangered** under Criteria E due to modelled declines in stream flow (Table 39). River and stream condition indices indicate that streams are under increasing stress, albeit relatively good condition based on some indices. However, assessments of the change in the functional features of the ecosystem over time suggest that decline in condition is highly likely.

Table 39. Summary of the IUCN Red List of Ecosystems assessment of the Australian *Alpine-subalpine Streams*.

Criteria	**A**	**B**	**C**	**D**	**E**	**Overall**
Sub-criterion 1 (past 50-years)	LC	LC	LC	VU	EN	EN
Sub-criterion 2 (50-year period including present & future)	DD	LC	LC	VU		
Sub-criterion 3 (since ~1750)	LC	LC	DD	DD		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria

Ecosystem Description

Ecosystem Classification

Australian *Alpine-subalpine Streams* aligns with F1.1 Permanent upland streams under the IUCN Global Ecosystem Typology v2.1 (Keith et al. 2022a). *Alpine-subalpine Streams* is not listed under Commonwealth legislation, but those in the Snowy River catchment in New South Wales have been listed as part of an endangered ecological community (*Aquatic Ecological Community in the Catchment of the Snowy River in NSW*) (FSC 2011). *Alpine-subalpine Streams* is classified according to the Strahler Stream Order Classification based on the number of tributaries linked to each stream (Strahler 1952, 1957); this ecosystem type includes first order (i.e., source streams with no other streams feeding into it), second order (i.e., where two source streams converge) and third order streams (i.e., where two second order streams converge).

Distinction from similar ecosystem types

Alpine-subalpine Streams flow through mountainous regions, typically at elevations above the tree line. Unlike non-alpine streams, they are often fed by snowmelt, have a shorter growing season, cooler temperatures, steeper gradients, and faster water flow (Campbell et al. 1986; Boulton et al. 2014). Alpine-subalpine Streams run through water dependent alpine riparian ecosystems, such as *Alpine Sphagnum Bog and Associated Fen*. These connected ecosystems are characterised by unique plant and animal species which have adapted to the more extreme conditions found at high elevations. Alpine wetlands such as *Alpine Sphagnum Bog and Associated Fen* are waterlogged areas which, like alpine rivers and streams, are fed by groundwater, precipitation and snowmelt, however they differ as they are not channels of flowing water (Boulton et al. 2014).

Distribution

Alpine-subalpine Streams occurs throughout the high-mountain regions of southern-eastern Australia (Australian Capital Territory, New South Wales, Victoria and Tasmania). In this assessment, Alpine-subalpine Streams incorporate all orders (*sensu* Strahler 1952) of permanently flowing water systems (including those that freeze seasonally) occurring above 1,500 m in the Australian Capital Territory and New South Wales, 1,370 m in Victoria, and 915 m in Tasmania (Costin 1957). The approximate geographical bounds of the ecosystem are 145.51° to 149.04° latitude and -35.44° to -43.53° longitude (Figure 96).

In total, *Alpine-subalpine Streams* covers an area of approximately 79 km^2 (Figure 96). Spatial products used to create this map represent the most accurate, national mapping available at the time of assessment (July 2021).

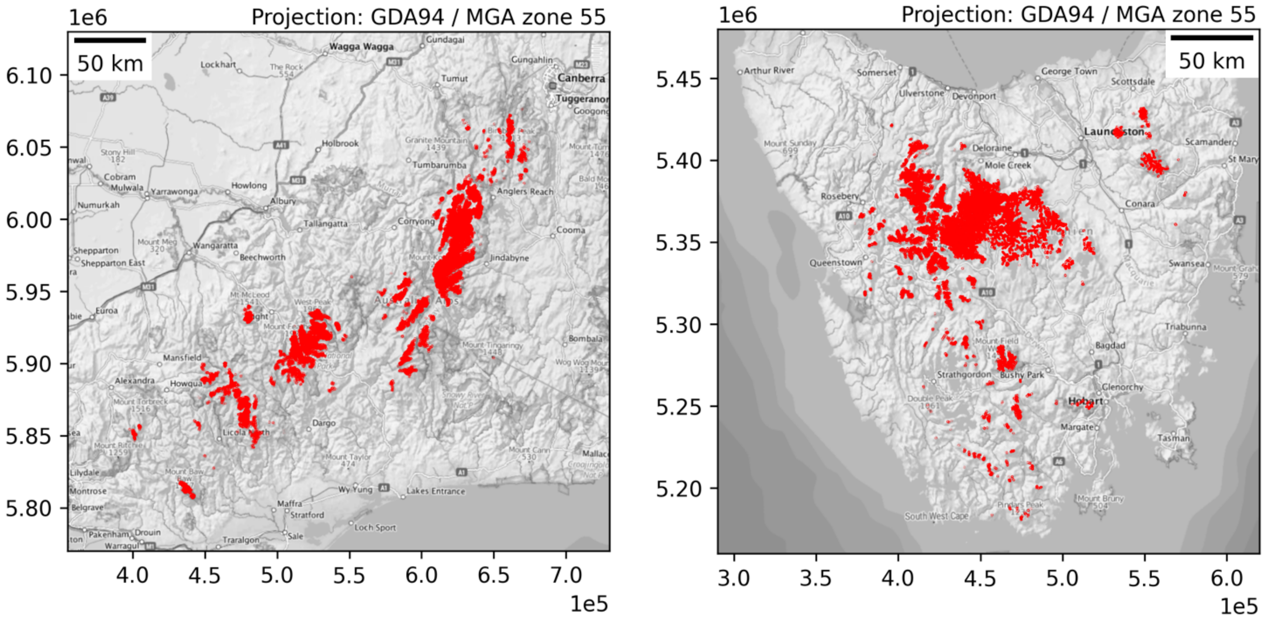


Figure 96. Distribution of *Alpine-subalpine Streams* (ref) above 915 m elevation in Tasmania (right), 1,370 m in Victoria and 1500 m in New South Wales (left).

Abiotic environment

Alpine-subalpine Streams in Australia typically occurs across high altitude regions and tends to have a unidirectional flow with a mean annual discharge of less than 5 GL (Campbell et al. 1986). The ecosystem type can vary in substrate from rocky to peat-like (Ponder et al. 1993) and has low mean flow, discharge, wash load, turbidity, and temperature. *Alpine-subalpine Streams* is characterised by shading by nearby vegetation, high solar inputs, retentiveness and substrate particle size (Boulton et al. 2014; Karis et al. 2016). Snow may act as an insulator during low winter temperatures, maintaining stream temperatures above 0 °C throughout the snow season. There are two common stream types; stony upland streams which had steep gradients and fast, turbulent flow, and less-steep plateau streams that have slower flow and include large pools with fine organic sediments (Campbell et al. 1986).

Alpine-subalpine Streams are typically fed by groundwater (Boulton et al. 2014). The water temperature and ionic content of these streams can vary depending on the influence of groundwater and surrounding vegetation (Clements et al. 2016). In general, these streams have low total dissolved solids (TDS; which decreases as altitude increases) and ionic proportions of $Na^+ > Ca^{2+} > Mg^{2+} > K^+$: $SO_4^{2-} > Cl^- > HCO_3^-$ (Silvester 2009; Chang et al. 2014). Thus, conductivity is usually low (Campbell et al. 1986). Dissolved oxygen is typically high (Campbell et al. 1986), and pH can be low in alpine and subalpine systems as a result of high dissolved organic carbon (Boulton et al. 2014). As streams move into subalpine regions, shading from riparian vegetation can reduce temperatures (Kalny et al. 2017) and diel range in temperatures, and changes can occur in ionic content due to surface runoff and rainfall (Hart & McKelvie 1985).

Characteristic native biota

There are no comprehensive studies of the fauna of high mountain streams in Australia. While knowledge of the vertebrate fauna is relatively extensive, knowledge of the composition and distributions of macroinvertebrates is relatively poor (Campbell et al. 1986).

Alpine-subalpine Streams has a high diversity of invertebrate species, with species richness increasing with stream size. On the mainland, streams at higher altitudes support fewer invertebrate species than streams lower altitudes (Campbell et al. 1986; Suter et al. 2002). *Alpine-subalpine Stream* macroinvertebrate fauna is dominated by insects, with the larvae of mayflies (Ephemeroptera), stoneflies (Plecoptera), caddisflies (Trichoptera), beetles (Coleoptera) and true flies (Diptera) all common and abundant (Campbell et al. 1986). Worms (Oligochaeta) and scuds or sideswimmers (Amphipoda) are common non-insects (Campbell et al. 1986). In Tasmania, species recorded include 17 species of stoneflies *(*Plecoptera), 13 mayflies (Ephemeroptera*),* 46 caddisflies (Trichoptera*)* (Dean & Cartwright 1992), 30 midges (Chironomidae), blackflies (*Simuliidae*), stoneflies (Plecoptera*)* and marsh beetles (Scirtidae) (Knott et al. 1978). Emerging aquatic insects use vegetation fringing streams, such as *Alpine-subalpine Open Grassy Heathland and Herbfield* and *Damp Valley Grassland and Rushland*. In Tasmania, *Alpine-subalpine Open Grassy Heathland and Herbfield* neighbouring *Alpine-subalpine Streams* is species- (and Tasmanian endemic) rich, with prominent species including *Caltha phylloptera*, *Ranunculus triplodontus*, and *Lilaeopsos polyantha* (Ponder et al. 1993). Common adaptations to alpine regions among insects include a reduction in wing size (brachyptery) or loss of wings (aptery), possibly as an adaptation to extreme thermal regimes and increased UV irradiation in alpine environments (Campbell et al. 1986). These adaptations would tend to reduce dispersal capability (Campbell et al. 1986).

Among these insects, there are 14 species of stonefly (*Plecoptera*) (e.g., *Thaumatoperla alpina, Dinotoperla marmorata, Austrocercella alpina*) appear restricted to above (or a small distance below) the treeline (Campbell et al. 1986). Similarly, the mayfly genus *Ameletoides* contains several species that are restricted to high mountain streams (Campbell et al. 1986). Caddisflies within the genera *Psyllobetina*, *Tanjilana* and *Ulmerochorema* are largely restricted to high mountain streams, as is Australia's largest known trichopteran, *Archaeophylax ocheus* (Campbell et al. 1986). The rare *Ramiheithrus virgatus* is a member of the family Philorheithridae and has only been collected from McKay Creek and Sassafras Gap in Victoria and Kosciusko Range in NSW (Campbell et al. 1986). The true fly family of net-winged midges (*Blephariceridae*) are highly characteristic of fast-flowing high mountain streams and there appear to be several species that are restricted to alpine regions including *Edwardsina gigantea*, *E. australiensis* and *E. torrentium* (Campbell et al. 1986).

Mountain shrimp and freshwater crayfish are also found in *Alpine-subalpine Streams*: *Anaspides tasmaniae* (restricted to the alpine streams on Mt Wellington) (Ahyong 2016), and *Euastacus reiki,* the only true alpine crayfish (Lawler & Crandall 1998). Other shrimp and crayfish species have wider distributions, occurring across a wide elevational extent of streams. The syncarid crustacean (*Anaspides tasmaniae*) has predominantly been collected above 750 m in Tasmania (Campbell et al. 1986). Amphipods comprise a substantial part of the invertebrate fauna of alpine streams. Gammarid amphipods from the genera *Austrogammarus* and *Neoniphargus* are (Campbell et al. 1986). Zooplankton are not common in *Alpine-subalpine Streams*, but *Ostracoda* and the copepod *Eucyclops* have been recorded in the spring-fed source pools (Clements et al. 2016).

Six species of galaxiid fish occur in mainland *Alpine-subalpine Streams*, including *Galaxias olidus*, *G. fuscus*, *G. gunaikurnai*, *G. mungadhan, G. tantangara* and *G. supremus* (Raadik 2014). Most of these species have highly restricted, high-elevation distributions, except for *G. olidus,* which ranges from the highest Australian peak (Mount Kosciuszko) to the coast (Raadik 2014). In Tasmania, a few freshwater fish species occur in *Alpine-subalpine Streams* associated with lakes, including Western paragalaxias (*Paragalaxias julianus*), Great Lake paragalaxias (*P. eleotrodies*) and Shannon paragalaxias (*P. dissimilis*). Many of the species are likely to have had more extensive distributions in the past, however, introduced brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchis mykiss*) outcompeted, preyed upon and severely fragmented populations, altering their distributions (Tilzey 1980; Raadik 2014).

Platypus (*Ornithorhynchus anatinus*) are rare above 1,500 m in Kosciuszko National Park but occur in streams on Mt Barrow, Tasmania (Ponder et al. 1993). Water skinks (e.g., *Eulamprus kosciuskoi,* alpine water skink) occur in riparian habitats at Davies Plain and Boggy Creek on the mainland, and the Gippsland water dragon (*Intellagama lesueurii howittii*) occurs along high elevation streams in Victoria (N. Clemann, pers. comm.). The alpine tree frog (*Litoria verreauxii alpina*) occurs near large, shallow alpine and subalpine pools on the mainland (Green & Osborne 2012), and *Crinia* spp. may also occupy streams with still, shallow pools where sun exposure can create warm pockets of water (D. Hunter, pers. comm.). Frog species including the plains brown tree frog (*L. paraewingi*) and the leaf green river tree frog (*L. nudidigita*) also occur in this ecosystem. Three subspecies of bango frog (*L. dumerilii*) occupy Victorian and Tasmanian alpine streams. Southern brown tree frog (*L. ewingii*) may occupy in alpine streams on the mainland and Tasmania, where pools of still water occur.

Alpine-subalpine Streams can support aquatic vegetation. In Tasmania, large sections of *Alpine-subalpine Streams* support few or no macrophytes (Ponder et al. 1993). Streams on the Central Plateau (Tasmania) support highly diverse macrophyte communities, with *Myriophyllum simulans* a dominant aquatic vegetation species (Ponder et al. 1993). Macroalgae – both filamentous and encrusting – are common in Tasmanian *Alpine-subalpine Streams*, but they remain poorly studied (Ponder et al. 1993). Source pools to alpine streams on the Bogong High Plains (Victoria) have a variety of mosses that are virtually restricted to this ecosystem, including *Blindia robusta, Polytrichum fluitans*, *Warnstorfia fluitans*, *Bartramia subsymmetrica* (listed as *B. bogongia*), *Pyrrhobryum mnoides*, and *Breutelia affinis* (McCartney et al. 2013; Clements et al. 2016). Three liverworts were recorded, including *Juggermannia orbiculate*, *Riccardia sp.* and *Isotachis montana* (McCartney et al. 2013; Clements et al. 2016). *Sphagnum novo-zealandicum* commonly occurs in *Alpine-subalpine Streams*, with the moss, *Andreaea australis,* and liverwort, *Heteroscyphus planiusculus,* present on in-stream rocks (Meaher & B. Fuhrer 2003).

Key components, processes, and interactions

Water temperature is a major variable determining the distribution, habitat use and life cycle of aquatic macroinvertebrates (Ward 1994; Fureder et al. 2001; Brittain 2008)(Figure 97). Temperature effects the hatching of eggs (Hynes & Hynes 1975; Suter & Bishop 1990; Brittain 1991, 1997; Brittain & Campbell 1991; Rotvit & Jacobsen 2014), nymph development and growth rates (Sweeney & Vannote 1978; Williams & Feltmate 1992), body size at emergence, fecundity (Brittain 1982, 2008; Williams & Feltmate 1992) and the length of the life cycle (Marchant et al. 1984; Brittain 2008). Water temperature also affects fish egg hatching, with galaxias at lower elevations having shorter incubation times than those at higher elevations (M. Lintermans, pers. comm). Any changes in thermal conditions in streams may affect the macroinvertebrate and fish communities present, particularly in *Alpine-subalpine Streams*, which are highly susceptible to climate change (New 2011).

Light penetration, solar energy and groundwater inputs all influence the temperature of *Alpine-subalpine Streams* (Figure 97). Light penetration is high as alpine streams are clear and not shaded by riparian vegetation and, except for deep winter snow, the bottom substrate receives solar inputs enabling virtually year-round growth of photosynthetic organisms (e.g., algae, bryophytes). In the subalpine zone, the presence of trees (e.g., *Eucalyptus pauciflora*, *E. stellulata*), fringing heath species and tea trees (*Leptospermum* spp.) may provide some shading. Solar inputs affect the temperature of *Alpine-subalpine Streams* with increased solar radiation increasing maximum water temperature (e.g., 4.7 to 22.8°C without shading, and 4.8 to 19.2°C with *E. pauciflora* present) and diurnal temperature variation (e.g., 8 to 14°C without shading at high elevations, and 2 to 4°C with shading at lower elevations) (Suter et al. 2011). Groundwater inputs at approximately 6ºC account for lower water temperatures and diurnal range at lower altitudes (Suter et al. 2011).

On mainland Australia, where snow accumulates and persists, stream temperatures remain above 0ºC throughout the snow season. With reduced snow cover, streams remain exposed to diurnal freeze-thaw cycles, which in turn affect stream temperatures and aquatic insects (Suter & McGuffie 2007). Without insulation from snow, increased frost crystal formation and frost heaving (the movement of either the ground or structures caused by the expansion of water in the soil as it freezes) occurs on stream banks, increasing soil erosion and runoff into streams (Good 1998)**.** Soil erosion and runoff also increase with high rainfall events.

In *Alpine-subalpine Streams*, water flow is important for transporting materials in solution compared with particulate organic debris at lower altitudes. The primary source of organic material in *Alpine-subalpine Streams* is dissolved organic carbon derived from peatlands (Silvester 2009; Karis et al. 2016), and smaller quantities from rainfall. Water flow also influences morphology and behaviour of many organisms, and indirectly determines habitat structure, water quality and ecology of interacting species (Boulton et al. 2014). Fast flowing riffle habitats with a substrate of large rocks, cobbles and pebbles support development of complex biofilms of algae, fungi, bacteria and detritus which, in turn, are grazed by invertebrates. Flowing water also provides particulate matter for filter feeders (Boulton & M. A. Brock 1999), and contributes to successful egg development in fish by maintaining relatively high oxygenation in riffle spawning sites and prevents smothering of eggs by fine sediment. During base flow, *Alpine-subalpine Streams* are frequently fast flowing, with stony substrates although plateau streams can be slower flowing with frequent pools and accumulated organic sediments (Campbell et al. 1986).

Sedimentation is a critical determinant of fish distributions, with all six species of mainland alpine-subalpine galaxias depositing demersal adhesive eggs on rocky substrates. There is no parental care of eggs in any of these species, and egg attachment and oxygenation are dependent on the presence of low silt (e.g., Stoessel et al. 2015).

Production of particulate organic matter either occurs within the stream itself (autochthonous) or from external inputs (allochthonous). However, energy sources driving *Alpine-subalpine Streams* are generally not well understood. *Alpine-subalpine Streams* flow through areas above the tree line, where the input of allochthonous organic matter is low. In these open-canopy streams, autochthonous sources of organic matter become more important where in-stream primary production dominates due to high light penetration driving photosynthesis (Closs et al. 2009). Thus, algal and in-stream plant production are potentially more significant than detrital processing and terrestrial inputs. In subalpine areas, where *E. pauciflora* stands are present, there is a change in the availability of food resources, with the potential for allochthonous resources to become more significant (Suter 2014).

Some macroinvertebrates can lack the enzymes required to degrade leaf litter and show preferential selection for leaf litter that has been colonized by aquatic Hyphomycetes (Suter 2014). However, macroinvertebrates will consume fine particulate materials in the water column (filter feeders), feed on accumulated particulate organic material (collector/gatherers), feed on biofilm (scrapers/grazers), consume large particulate particles such as leaves and the biofilm on them (algae shredders), and eat other invertebrates (predators) (Boulton et al. 2014).

Photosynthesis in the alpine stream occurs via algae and bryophytes (mosses and liverworts), but their biomass is limited by low nutrient concentration and low temperature of the streams.

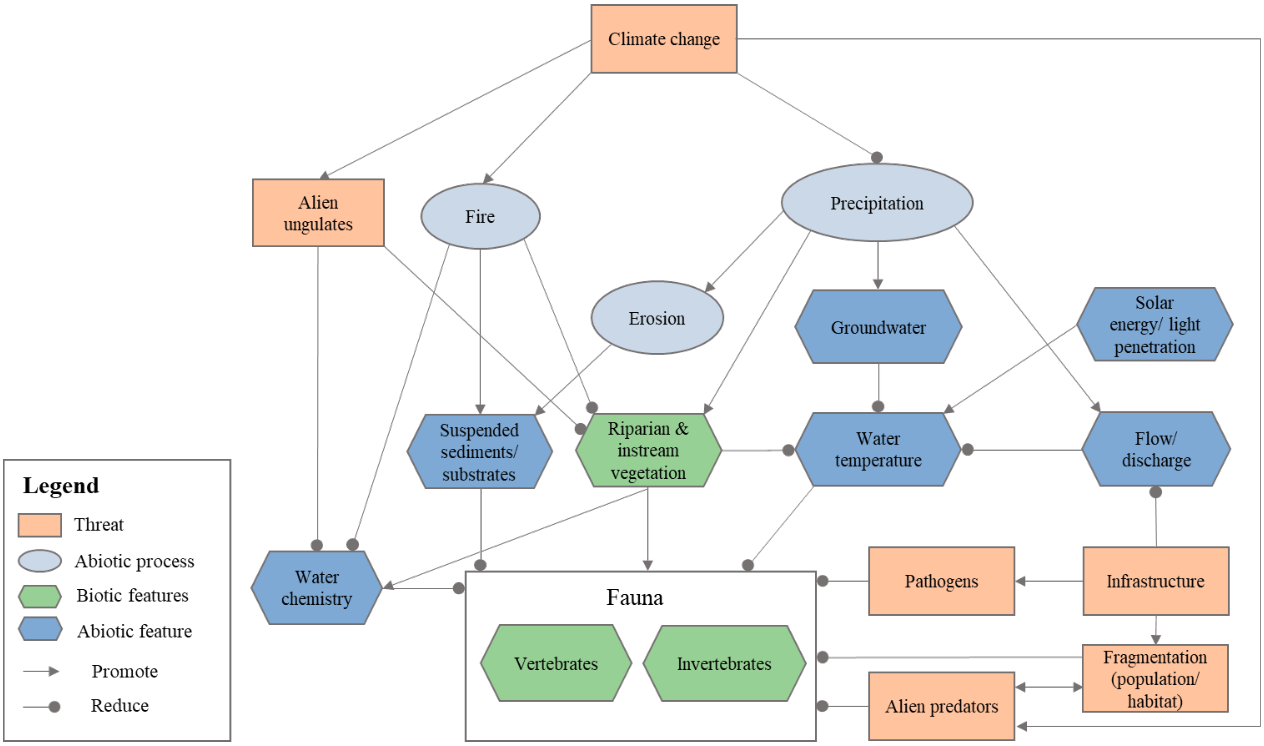


Figure 97. Conceptual diagram illustrating relationships between, and threats to, biotic and abiotic components and processes in Alpine-subalpine Streams.

Major threats

Climate change

Climate change poses a major threat to *Alpine-subalpine Streams* via predicted impacts on hydrological regimes, water quality and ecological processes (Figure 97). Increased temperatures have already reduced snow area, depth and persistence in Australian alpine and subalpine ecosystems, with this impact expected to increase with continued temperature increases (Slatyer 2010). This will likely lead to more frequent and extreme events such as floods and droughts, and changing seasonal patterns of stream flows (Love et al. 2019). A drying climate may increase the intermittency of alpine streams and thus alter the abundance and diversity of macroinvertebrates (Siebers et al. 2020; Drost et al. 2022) by exposing them to higher temperatures, evaporation and salinity. Changes in freeze-thaw cycles are associated with changes in channel stability, temperature and nutrient loadings which lead to complex ecological shift in stream communities, food webs and biodiversity (Brighenti et al. 2019).

Climate change is also predicted to cause lower-altitudinal species to shift their distribution to higher altitudes, although there are few studies that quantify this in Australian Alpine freshwater ecosystems (Hughes 2003). Elevational migrants (e.g., plants, animals, pathogens) are species that move up from lower altitudes due to climate change or other factors and compete with, predate on, or displace endemic alpine species (Khamis et al. 2015).

Pollution from fires and fire-fighting activities in nearby ecosystems is a threat to *Alpine-subalpine Streams*. The chemical properties of streams can be altered by ash and flame retardants, which can increase the salinity (Gimenez et al. 2004; Kalabokidis 2018). Burnt peat can release nitrogen into the water, altering the nutrient balance (Van Beest et al. 2019). Burnt vegetation reduces the shade, increasing the solar radiation exposure and potentially increasing algal blooms (Klose et al. 2015). Rainfall after a fire can deliver fine and coarse sediment into the stream, resulting in a reduction of short-term water quality particularly within the first-year post fire in subalpine areas close to the fire perimeter (Reneau et al. 2007). Longer term impacts associated with persistent sand slugs resulting from post-fire erosion can smother in-stream habitats and leading to large reductions in fish assemblages (Boulton et al. 2014). Additionally, the removal of vegetative ground cover by fire can make soils hydrophobic, leading to increased overland flow, peak flow, and erosion, which deposits ash and sediment into streams (Prosser & Williams 1998; DeBano 2003; Shakesby & Doerr 2006). Post-fire catchment erosion can deposit large boulder and cobble fans, altering channel morphology and gradient with long-lasting sand and gravel "slugs" that reduce habitat complexity and heterogeneity (Bartley & Rutherfurd 2005; Short et al. 2015). Filter feeding aquatic invertebrates are also known to reduce post-fire (Anon 2003; Suter & McGuffie 2007).

Invasive and introduced species

Trout are not native to Australia and were introduced from Europe and North America for recreational fishing. They have since spread to many alpine streams and rivers where they now threaten endemic species such the Alpine Stonefly (*Thaumatoperla alpina),* which is listed as endangered under the EPBC act (Threatened Species Scientific Committee 2011). Additionally, invasive fish species such as trout pose a significant threat to aquatic ecosystems as they can outcompete and prey on native fish such as galaxias (McIntosh et al. 2010; Woodford & McIntosh 2013). However, natural barriers such as swampy areas with diffuse or subterranean flow, waterfalls, and artificial barriers such as aqueducts can limit their spread upstream.

Invasive ungulates, such as fallow and sambal deer, feral horses and cattle, cause damage to alpine streams through browsing and grazing, trampling and pugging. Their selective grazing patterns lead to changes in the composition of vegetation along stream banks, which can have cascading effects on the ecosystem – grazing increases sedimentation in streams and in turn alters the composition of vegetation along stream banks (Robertson et al. 2019).

Isolation and fragmentation

Land clearing, infrastructure development, and natural disturbances threaten *Alpine-subalpine Streams* by reducing and fragmenting the ecosystem type. Isolation/fragmentation can affect the diversity and distribution of aquatic macroinvertebrates, which are important indicators of stream health and function (Monaghan et al. 2005). Fragmentation by dams or reservoirs can disrupt the connectivity and continuity of stream habitats, altering their hydrological regime, water quality, sediment transport and temperature. This can create barriers for dispersal and migration of macroinvertebrates, leading to reduced gene flow, increased inbreeding and loss of genetic diversity (Monaghan et al. 2005).

Infrastructure development

Construction of hydro-energy storage poses a significant threat to alpine streams, including impacts on biodiversity, introduction of invasive species, altered hydrology, and reduced water quality (Gabbud & Lane 2016; Normyle & Pittock 2020). For example, the Snowy 2.0 dam development on the mainland will involve significant tunnelling (through 27 km of rock) and lead to a significant decrease in environmental flow in certain areas; Snowy 2.0 may also introduce new non-native species into some catchments. For example, in Victoria, for rivers and streams not part of the protected area network, construction of connectors and canals associated with the Snowy 2.0 scheme will threaten alpine streams by linking previously disconnected areas, facilitating dispersal of predatory invasive fish and disease such as chytrid fungus or rock snot.

Pathogens

Fishing activities may facilitate the spread of fungi including chytrid (*Batrachochytrium dendrobatidis* and *mucor amphibiorum)*, and freshwater algae (rock snot), although this has yet to be observed. Anglers and the Snowy 2.0 development (via artificial connectors and aqueducts) may increase the risk of spreading Epizootic Haematopoietic Necrosis (EHN) virus to *Alpine-subalpine Streams* from lower elevations streams. The virus is known to affect the native species *Galaxias olidus* and the introduced rainbow trout (which may be a carrier), however the impact on other galaxid fish species is unknown (Langdon 1989).

IUCN Stresses Classification

1.2 – Ecosystem degradation (Altered temperature/precipitation/fire regime)
2.3.2 – Indirect species effects (Competition)

IUCN Threats Classification

6.1 – Recreational activities
7.2 – Dams and water management/use
7.7.1 – Increase in fire frequency/intensity
8.1 – Invasive non-native/alien species/diseases
8.1 – Non-native/alien species/diseases
11.1 – Habitat shifting & alteration
11.3 – Temperature extremes
11.5 – Climate change & severe weather; other impacts

Ecosystem collapse

Based on the processes and threats driving the system, *Alpine-subalpine Streams* may have four collapse states:

1) If the hydrological conditions change from permanent to ephemeral or seasonal. This may occur via a reduction in the surface or groundwater, or reduction in precipitation under climate change to the point where the streams are unable to persist year-round. This may occur with a minimum of one dry period within a year, representing a transition to a novel ecosystem type.
2) If there was a significant reduction in water availability so there was not a visible water table, resulting in total loss of any permanency of water in streams.
3) If the composition of the aquatic biota changed to being dominated by non-native, invasive species.
4) If there was a loss of characteristic wet-affiliated species (e.g., *Empodisma*), indicating that chemical properties of the steam are no longer suitable, or the stream is not sufficiently wet to support these species.

Ecosystem collapse in *Alpine-subalpine Streams* is defined in this assessment as when:

1. Area: The mapped distribution of streams with permanent water declines to zero (100% loss) (Criteria A and B).
2. Annual baseflow: The annual baseflow component of the hydrograph declines to 0 m^3/s (Criterion C).
3. Macroinvertebrate assemblage: The SIGNALT macroinvertebrate assemblage condition score is 3.4 in Victoria, 3.1 in Tasmania and 1.9 in NSW and ACT (Criterion D).

4. Length of cease-to-flow periods: The median length of cease-to-flow period > 30 days (or maximum of 90 days) in the median future climate projection and noting where those thresholds are crossed in the 90th percentile projection (Criterion E).
5. Length of low flow spells: The median length of low flow spells is > 60 days (or maximum of 120 days) in the median future climate projection and noting where those thresholds are crossed in the 90th percentile projection (Criterion E).

Risk Assessment

Criterion A: Reduction in distribution

Summary

There may be future reductions in the distribution of *Alpine-subalpine Streams* due to future climate change reducing the availability of water (Lough & Hobday 2011) (sub-criterion A2). Past changes in distribution have not been quantified but appear very unlikely to have exceeded a 30% reduction in the past 50 years (sub-criterion A1) or since European invasion (sub-criterion A3). Hence, the status of this ecosystem is likely to be **Least Concern** based on sub-criteria A1 and A3, while sub-criterion A2 is **Data Deficient**.

Criterion B: Restricted distribution

Summary

The Extent of Occurrence (EOO) for *Alpine-subalpine Streams* is **Least Concern** under Criterion B.

Methods

Under criterion B, the Extent of Occurrence (EOO; B1) and Area of Occupancy (AOO; B2) of *Alpine-subalpine Streams* were determined using an existing national map product (National Surface Water Hydrology: Crossman & Li 2015; Geoscience Australia 2021). A complete description of the data and methods used to create the current distribution map for this ecosystem is provided in the main methods. The EOO was calculated using a minimum convex polygon enclosing all mapped occurrences of alpine-subalpine streams in Australia. The AOO was calculated based on the number of 10 x 10 km grid cells that contained the ecosystem. The number of threat-defined locations was based on fire as this is the most important stochastic threat to alpine-subalpine streams.

Assessment outcome

Sub-criterion B1

The Extent of Occurrence (EOO) of *Alpine-subalpine Streams* is currently estimated at 161,734.52 km^2 (Figure 98). The risk status is **Least Concern** under sub-criterion B1.

Sub-criterion B2

Alpine-subalpine Streams occupies 276 10 x 10 km grid cells (Area of Occupancy, AOO) (Figure 98). While there is evidence that the ecosystem may decline due to documented reductions in snow persistence, the AOO is well above the threshold for Vulnerable. Thus, the risk status of the ecosystem is **Least Concern** under criterion sub-B2.

Sub-criterion B3

The main threats to *Alpine-subalpine Streams* are exotic invasive plant and animal species outcompeting native species, changes in air/water temperature altering the water chemistry and availability due to changes in snow melt times, and run-off from fires polluting the water. However, these threats occur locally across the distribution at different times and therefore the number of threat-defined locations is more than 5. These threats are unlikely to cause collapse of the ecosystem within a short timeframe (~20 years). Therefore, the risk status of the ecosystem is **Least Concern** under sub-criterion B3.

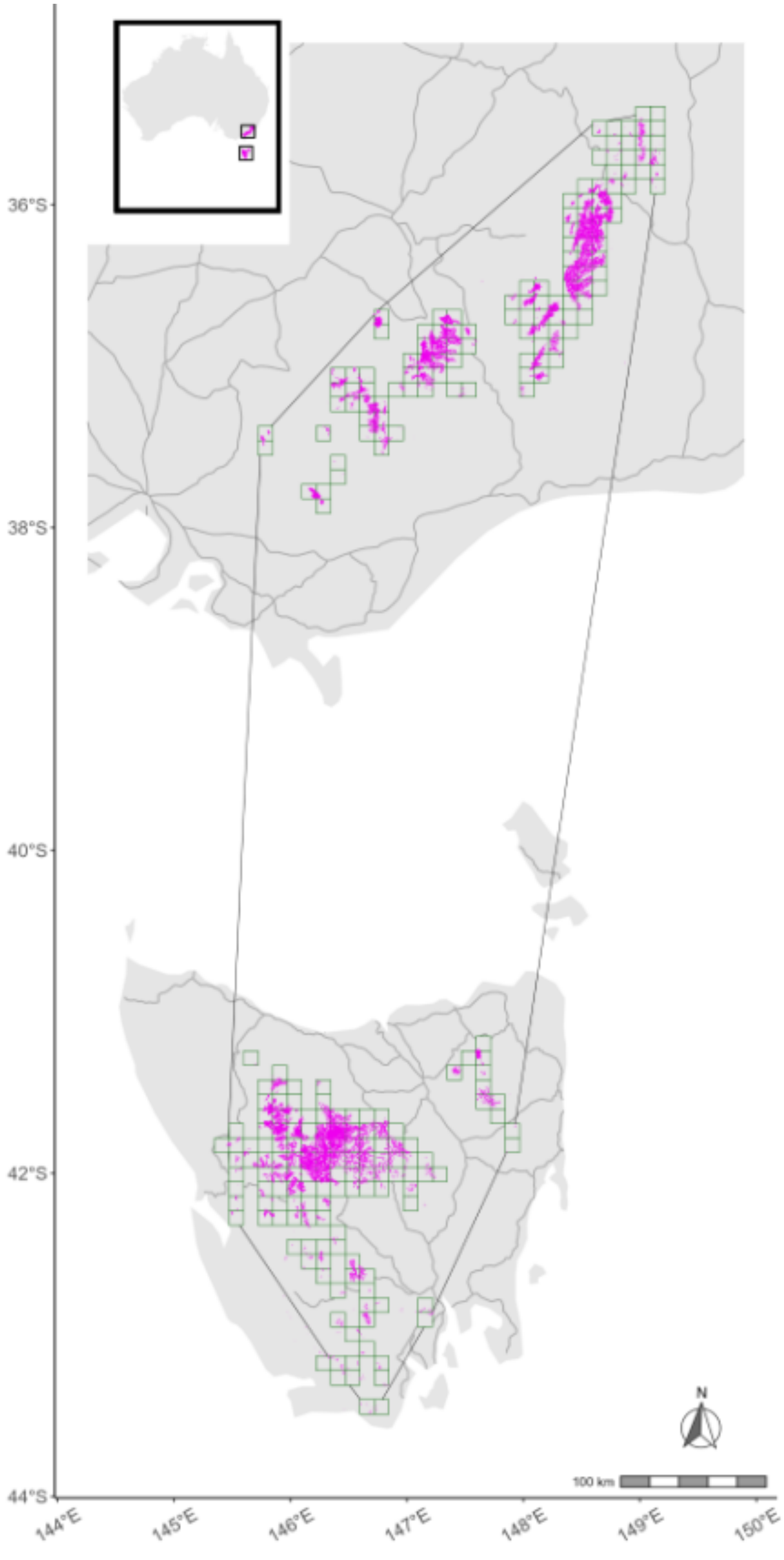


Figure 98. Map of Alpine-subalpine Streams (magenta polygons), showing EOO (black polygon) and AOO (green squares).

Criterion C: Environmental degradation

Summary

Based on the available temporal and spatial hydrological data for *Alpine-subalpine Streams*, there were no identified trends in annual baseflow in the recent past, while insufficient data exist for long-term (historical) analyses. Therefore, the status of the ecosystem is **Least Concern** under sub-criteria C1 and C2b and **Data Deficient** under sub-criterion C3.

Identification of abiotic indicator

We examined the relevance and data availability/quality for three abiotic indicators to assess the risk of collapse from environmental degradation:

- *Annual baseflow*: a measure of the volume of water that represents the contribution of groundwater to a stream.
- Water temperature: as a measure of the temperature (°C) of the waterflow
- Fire occurrence: a measure of the occurrence of fires adjacent to streams

Indicator: Annual baseflow

Relevance to ecosystem function

Alpine-subalpine Streams are defined as being permanently flowing systems. This is associated with the presence of a water table above the surface. Changes to the water may lead to the system transitioning from permanent to intermittent or ephemeral streams, or to a lack of any permanent standing (or flowing) water (Saft et al. 2020). An accepted definition of a 'permanent or near permanent' stream is one that experiences predictable flooding, where the annual input of water is greater than the losses in 9 out of 10 years, or one that is predictably filled, with annual inflow exceeding minimum annual loss 90% of the time (Boulton et al. 2014). While near permanent streams can dry during extreme drought, any increase in the regularity of that drying, or shift to seasonal or frequent drying fundamentally changes the character of the stream, potentially reducing the suitability of *Alpine-subalpine Streams* for many characteristic species as these systems often support diverse aquatic life, much of which cannot tolerate desiccation (Boulton et al. 2014). During dry conditions, the majority (or all) of flow in a stream is derived from groundwater. As a result, the amount of baseflow in the system is directly related to the continued status of *Alpine-subalpine Streams* as near permanently flowing systems. If baseflows decline through time, that suggests that the stream is drying and is more likely to be intermittent or ephemeral.

Data availability and quality

Recent change (C1)

We identified the locations of all available streamflow gauges available from the Bureau of Meteorology (BOM 2025) using the R package 'hyrdogauge' (Holt 2025). We then used the R package 'geodata' to intersect those locations using the Shuttle Radar Topography Mission (SRTM) layer (CGIAR-CSI 2018) to identify those that fell above the elevation threshold for alpine/subalpine areas in the relevant state. Where there were fewer than five gauges in the alpine/subalpine area (e.g., in ACT), we relaxed the elevation threshold to include a minimum of five gauges in each state (these are identified where relevant). Five was selected as a common minimum sample size used to detect relationships. As a result, we had 13 gauges above the elevation threshold across NSW, Victoria and Tasmania, with an additional 12 gauges in ACT, NSW and Victoria that fell below the elevation threshold but were included in a second analysis to assess the impact of having a small number of gauges that were truly alpine. We assumed that these gauges were representative of flows in alpine/subalpine streams across the four states.

Ten of the 13 alpine/subalpine gauges had records that extended for at least 50 years. For ease of analysis, all years of record were included for all 13 gauges.

Data quality was variable. Some gauges had data that depart markedly from the patterns observed at other nearby gauges, which was indicated by quality codes indicating poor quality data (i.e. >150). These gauges were excluded, leaving 11 gauges in the alpine-subalpine region and 23 in the larger data set (Figure 99).

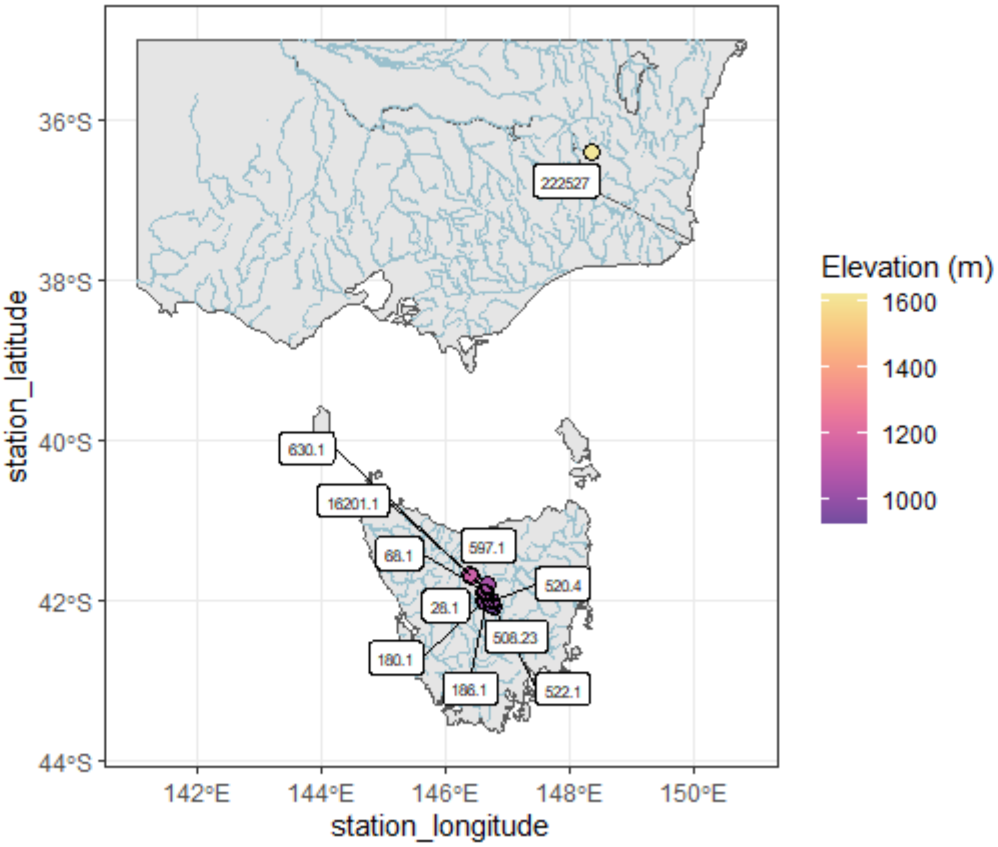


Figure 99. Gauge locations used to assess trends in baseflow over the historical record including 11 gauges in the alpine/subalpine zone across three states. Labels show gauge ID.

Future change (C2)

We extrapolated the trend in flows to estimate the trajectory of decline for *Alpine-subalpine Streams* across the two sets of gauges (13 alpine/subalpine and 23 to include at least 5 in each state).

Historical change (C3)

Data were unavailable to assess historical change in hydrologic indicators.

Selection of collapse threshold

The drying of *Alpine-subalpine Streams* will have a large impact on the character of those streams as many freshwater biota are restricted to alpine streams and wetlands because they cannot tolerate warming or drying (Boulton et al. 2014). Given this intolerance to drying, we determined that a baseflow of 0 would constitute collapse. Thus, we define collapse as when the baseflow component of the hydrograph reached 0 m^3/s.

Calculation of initial and present/future values

Recent change (C1) and Future change (C2)

Baseflow was calculated for each year in the historical time series for each gauge using the R package *hydrostats* (Bond 2022). Baseflow varies among stream types, tending to be more stable in larger streams with near-continuous connections to groundwater and varying more in smaller streams that are intermittently connected to groundwater, usually in wetter years, for example.

We fit a generalised linear model to the data, assessing baseflow as a function of year, with gauge included as a random variable. There was no statistically significant trend in baseflow across the suite of gauges through time (**Error! Reference source not found.**100) or between states, and so no extrapolation was undertaken to assess future change.

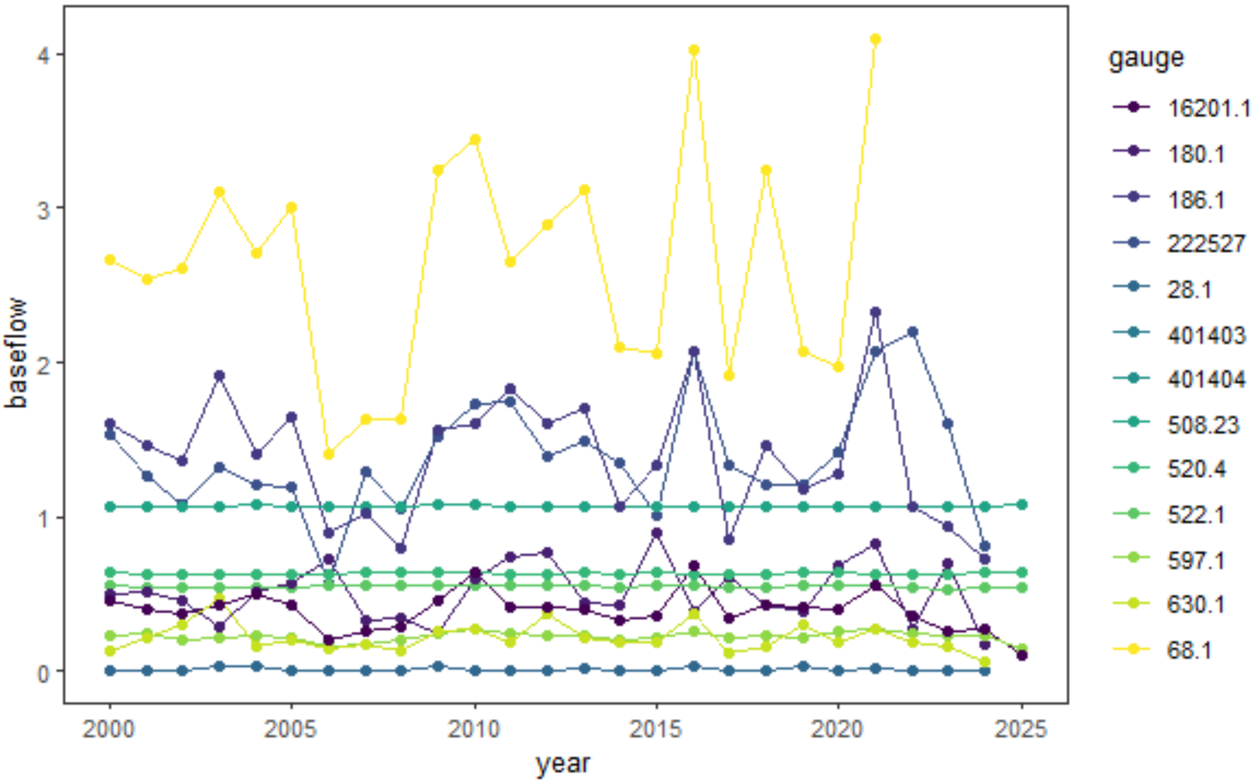


Figure 100. Trend in baseflow over the historical record for 11 gauges in the alpine/subalpine zone across three states, illustrating the pattern between 2000 to 2025 (noting that the period of data for each gauge varies). Analyses were conducted on the available length of record for each gauge (range 1922-2025).

Historical change (C3)

Data were unavailable to assess this sub-criterion.

Calculation of severity and extent

Recent change (C1) and Future change (C2b)

There was no trend in baseflow across the historical record. As a result, the risk status is **Least Concern** under sub-criterion C1 and sub-criterion C2b.

Historical change (C3)

There were insufficient data to assess historical changes in baseflow. The risk status is **Data Deficient** under sub-criterion C3.

Indicator: Water temperature

Relevance to ecosystem function

Water temperature is a key factor for the aquatic biota present in *Alpine-subalpine Streams* as it impacts the timing of invertebrate breeding and provides an ecological barrier that allows endemic species to outcompete invasives (i.e. because invasive species are often unable to tolerate the cold temperatures). Stream water temperature can be largely driven by groundwater temperatures, although is also influenced by light penetration and air temperature (Meier et al. 2003; Brown et al. 2007). The characteristic biota of *Alpine-subalpine Streams* cannot tolerate warming (Boulton et al. 2014). As such directional warming of stream temperatures, particularly during winter would constitute a decline in condition in *Alpine-subalpine Streams*, risking a loss of those characteristic taxa.

Data availability and quality

Recent change (C1)

We identified the locations of all available gauges with temperature records available from the Bureau of Meteorology (BOM 2025) using the R package ‘hydrogauge’ (Holt 2025). As for the streamflow records, we then used the R package ‘geodata’ to intersect those locations using the Shuttle Radar Topography Mission (SRTM) layer (CGIAR-CSI 2018) to identify those that fell above the elevation threshold for alpine/subalpine areas in the relevant state. No gauges measuring temperature were found in above the threshold for alpine/subalpine regions in any state and so we relaxed the elevation threshold to include a minimum of five gauges in each state (where these fell in similar locations to the alpine regions). Five was selected as a common minimum sample size used to detect relationships. As a result, we had 14 gauges below the elevation threshold across ACT, NSW, Victoria and Tasmania (Figure 101). We assumed that these gauges were broadly representative of temperatures in alpine/subalpine streams across the four states, but perhaps somewhat higher due to their lower elevations.

Only one gauge had records that extended for at least 50 years. For all analyses, all years of available record were included for each gauge.

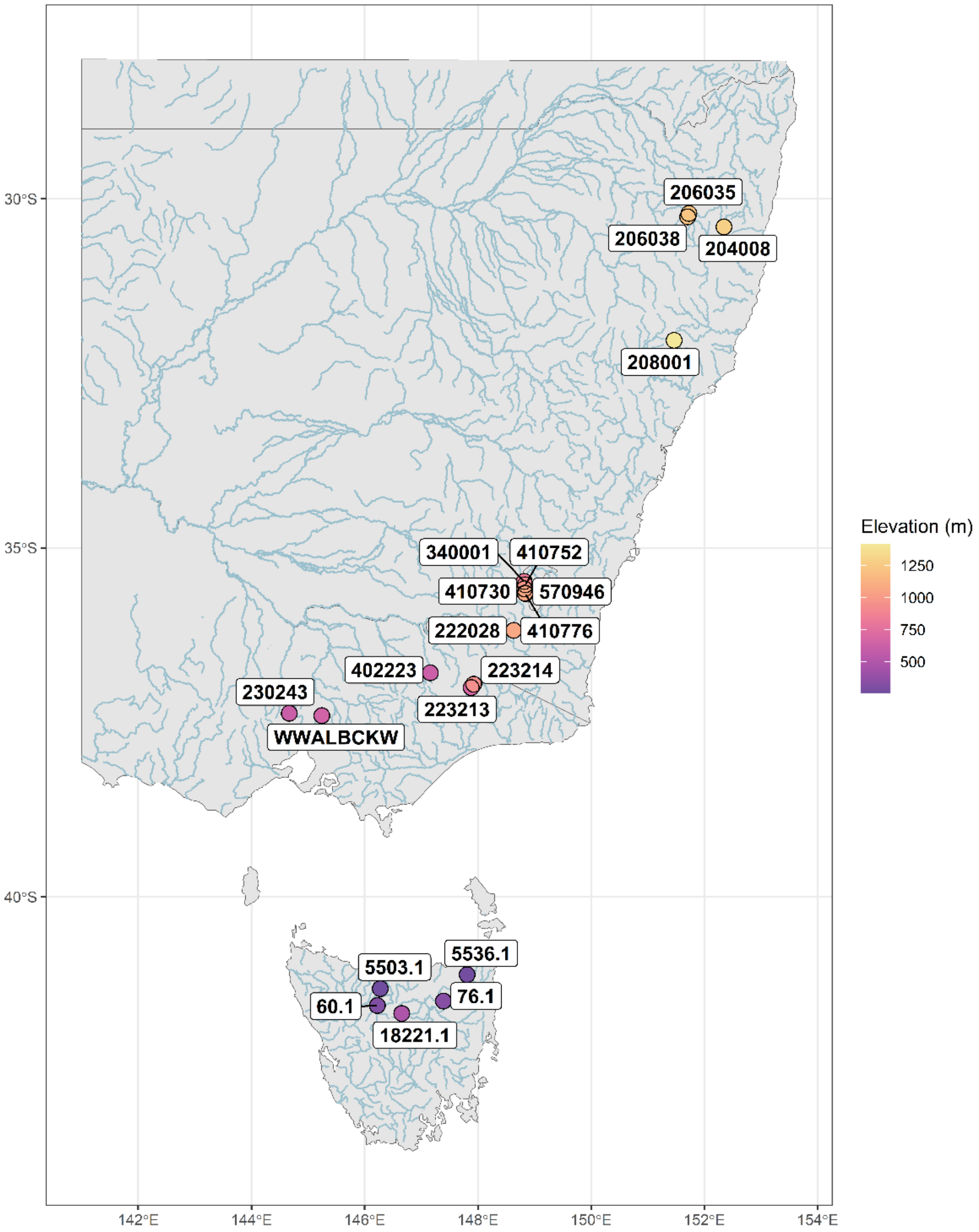


Figure 101. Gauge locations used to assess trends in temperature over the historical record including 14 gauges below the alpine/subalpine zone across four states, noting that no gauges existed above the elevation thresholds in any state.

Future change (C2)

We extrapolated the trend in temperature to estimate the trajectory of warming for *Alpine-subalpine Streams* across the available gauges.

Historical change (C3)

Data were unavailable to assess historical change in temperature.

Selection of collapse threshold

The warming of *Alpine-subalpine Streams* will have a large impact on the character of those streams as many freshwater biota are adapted to cold temperatures and cannot tolerate warming

(Boulton et al. 2014). Given this intolerance to warming, we determined that an average annual surface water temperature of 20ºC would constitute collapse, or an average winter temperature that exceeds 5ºC.

Calculation of initial and present/future values

Recent change (C1) and Future change (C2)

Average and minimum annual temperatures were calculated for each year in the historical time series for each station using the R statistical environment, along with average seasonal temperatures. Because of the very large variance in the duration and magnitude of long-term average temperatures (Figure 102), we standardised all average temperatures by the long-term average temperature for that stream (Figure 103). We also removed any stations that had fewer than 15 years of record, as long-term averages would not be meaningful. This resulted in a sample size of 8 stations, which is small and so results from these analyses should be considered with caution, particularly given that all gauges are also below the elevation thresholds in each state. We fit a generalised linear model to the data, assessing average annual temperature as a function of year, with a station by year interaction included as a random variable. We repeated this analysis for average winter temperature (Figure 104) standardised by the long-term average winter temperature.

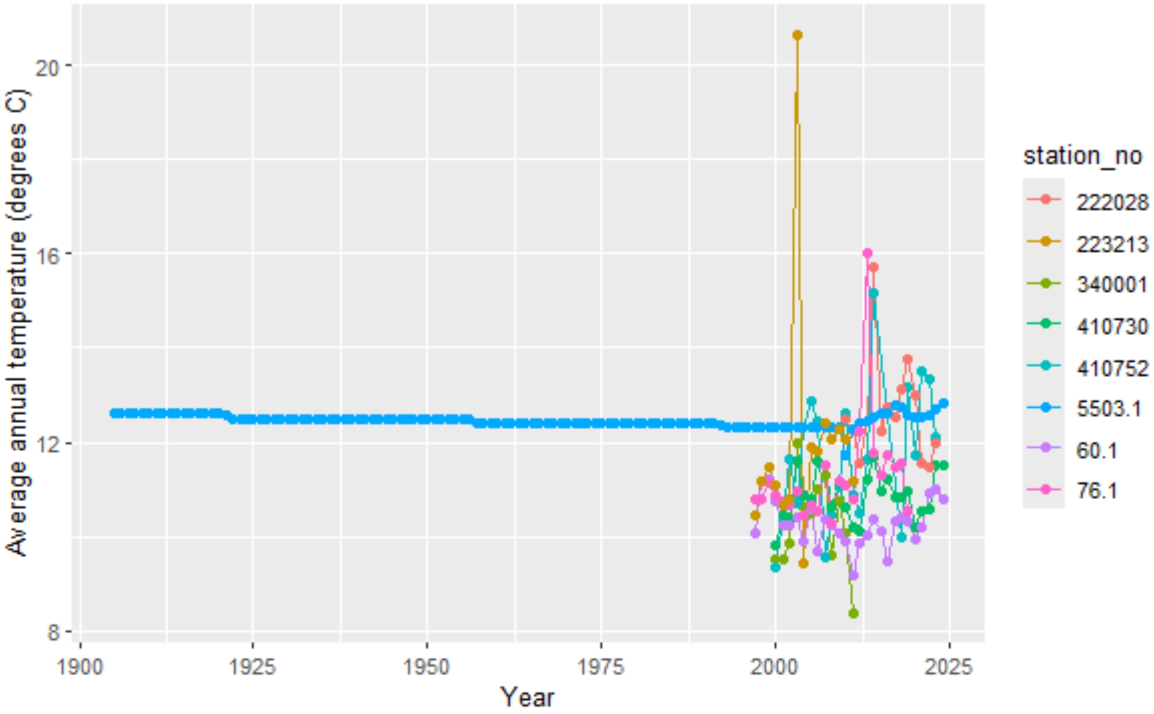


Figure 102. Trend in average annual temperature over the historical record for 8 gauges below the alpine/subalpine zone across four states (noting that the period of data for each gauge varies). Analyses were conducted on the available length of record for each gauge (range 1904-2025).

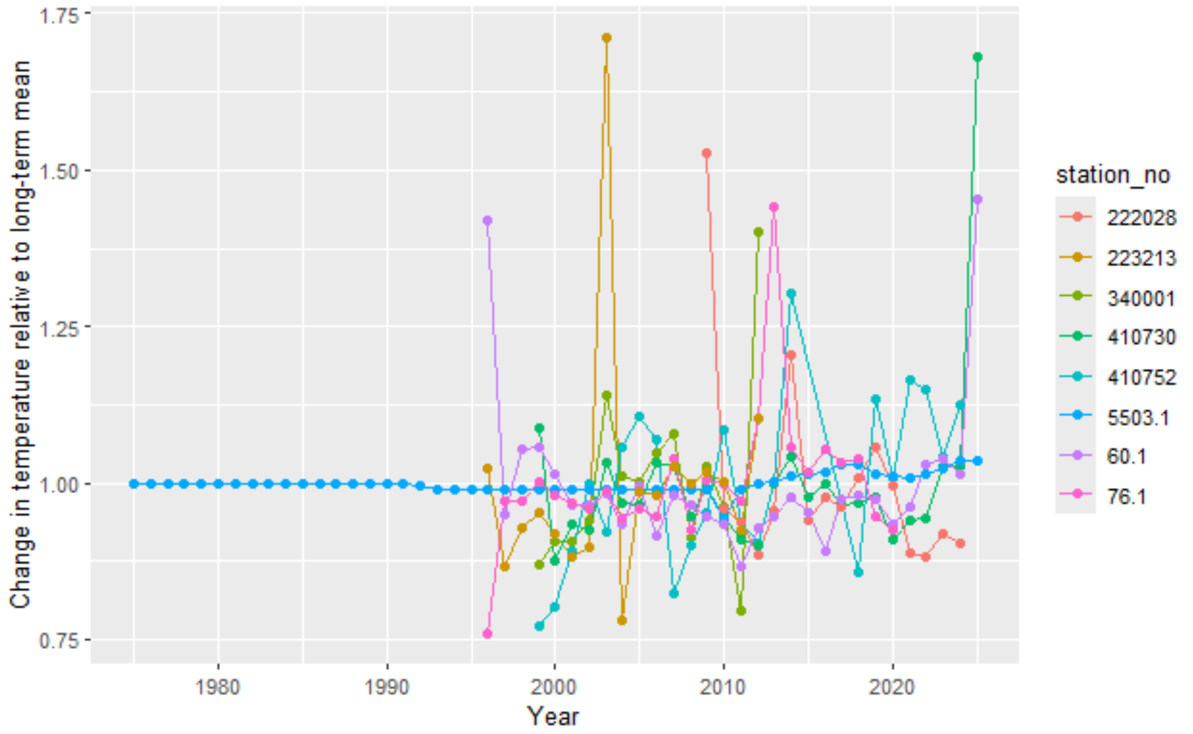


Figure 103. Trend in change in average annual temperature relative to the historical mean for 8 stations that had more than 15 years of record, expanded below the alpine/subalpine zone across four states, illustrated from 1975 (noting that the period of data for each station varies). Analyses were conducted on the available length of record for each station, where that station had more than 15 years of record (range 1904-2025).

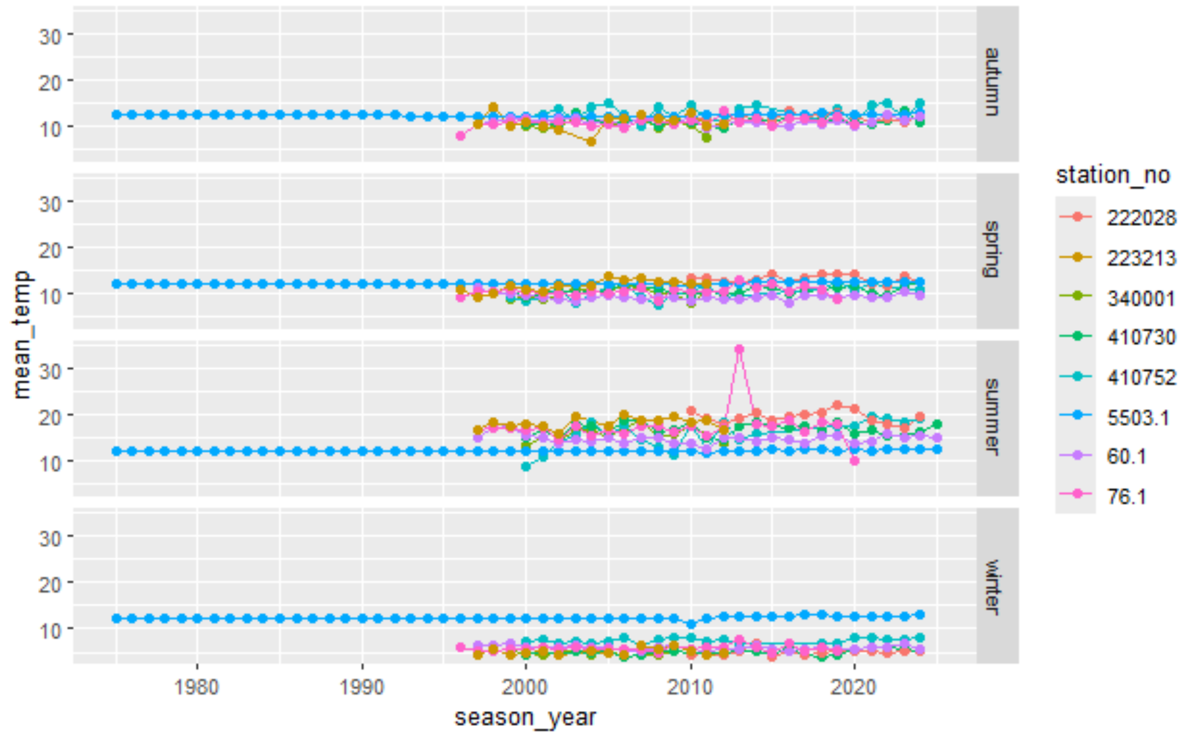


Figure 104. Trend in average seasonal temperature over the historical record for 8 gauges below the alpine/subalpine zone across four states, illustrated from 1975 (noting that the period of data for each gauge varies). Analyses were conducted on the available length of record for each gauge (range 1904-2025).

Historical change (C3)

Data were unavailable to assess this sub-criterion.

Calculation of severity and extent

Recent change (C1) and Future change (C2b)

There was a statistically significant increase in relative annual average temperature through time ($F_{1,203}$ = 3.03). Similarly, there was a statistically significant increase in relative average winter temperatures through time ($F_{1,203}$ = 1.91) (Figure 105).

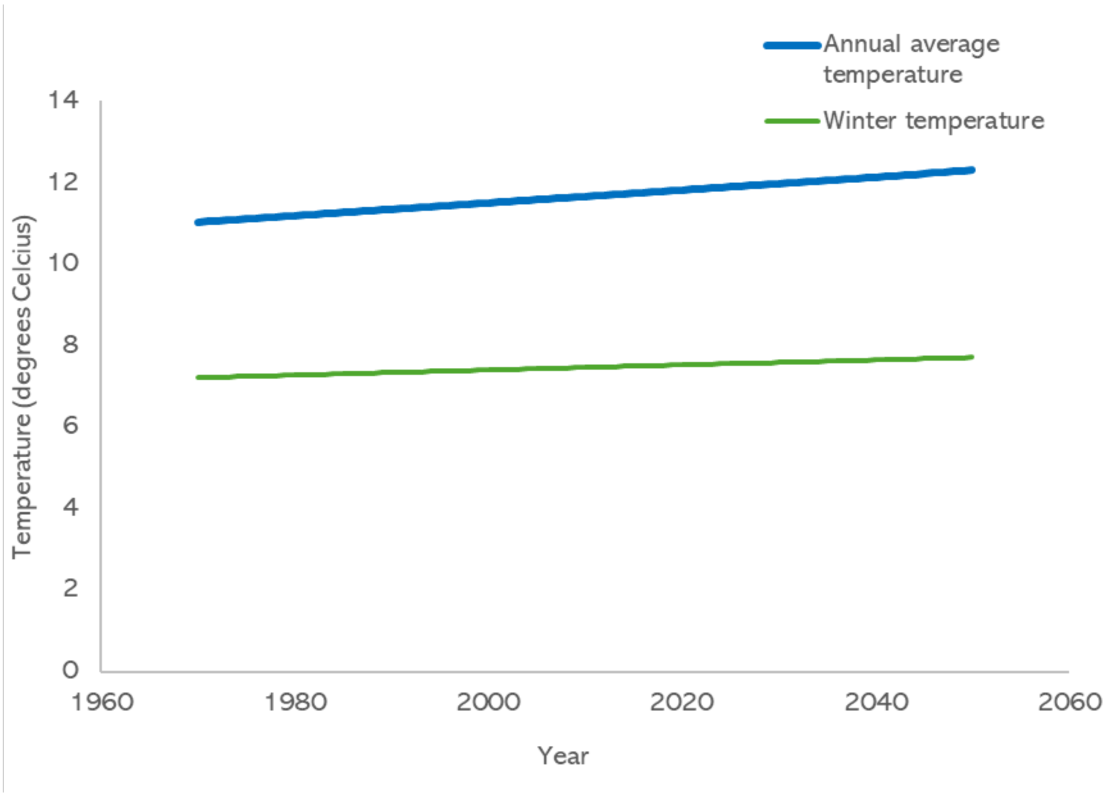


Figure 105. Trend in average annual and winter temperatures extrapolated for the period of 1970 to 2050. Analyses were conducted on the available length of record for each sampling location. The relevant collapse thresholds (of 15 °C for winter and 20 °C for average annual temperature) are beyond the scale of the y axis.

To calculate severity of change, we calculated the average change per year in temperature over a 50-year period, based on the calculated trend.

For the period 1970-2020 (relative to sub-criterion C1), average annual temperature had changed from an estimated 10.72 to 11.20°C, equating to a relative severity of 100 x (10.72°C – 11.20°C)/(10.72°C – 20°C) = 5.17%. For the period 2000-2050 (relative to sub-criterion C2b), average annual temperature was predicted to change from to 12.30°C, equating to a relative severity of 100 x (1.51°C – 12.30°C)/(1.51°C – 20°C) = 17%.

For average winter temperatures, for the period 1970-2020 (relative to sub-criterion C1), average temperature had changed from an estimated 7.10 to 7.28°C, equating to a relative severity of

100 x (7.10ºC – 7.28ºC)/(7.10ºC – 5ºC) = -8.57%. For the period 2000-2050 (relative to sub-criterion C2b), average winter temperature was predicted to change from 7.40 to 7.71ºC, equating to a relative severity of 100 x (7.40ºC – 7.71ºC)/(7.40ºC – 5ºC) = -12.92%.

Thus, the risk status under sub-criteria C1 and C2b was **Least Concern** based on this indicator.

Historical change (C3)

There were insufficient data to assess historical changes in baseflow. The risk status is **Data Deficient** under sub-criterion C3.

Indicator: Fire

Fires in ecosystems adjacent to *Alpine-subalpine Streams* can degrade the water quality. Run-off containing pollutants from the burnt vegetation may change the water chemistry and increase sedimentation, affecting the native biota by varying the dissolved oxygen (Dahm et al. 2015). The water turbidity may also increase, reducing light penetration and thus reducing water temperature in alpine streams (Chen & Chang 2022). However, in subalpine streams, the riparian canopy cover may be reduced post-fire leading to increased temperatures from higher solar radiation exposure. Further, the elevated use of groundwater as vegetation regrows postfire may reduce the groundwater level and slow stream recharge (Filoso et al. 2017). However, there is insufficient information to understand the relationship between what frequency or severity of fires impacts on stream water quality or biota. Therefore, the ecosystem is **Data Deficient** for this indicator.

Criterion D: Disruption of biotic processes and/or interactions

Summary

Based on the available macroinvertebrate assemblage condition data for *Alpine-subalpine Streams*, there was a statistically significant decline in condition over the historical record in Victoria, NSW and Tasmania (noting there were insufficient records available for ACT). Over a 50-year period, this decline represented a change of 40-50% relative to the collapse threshold. Therefore, the status of the ecosystem is **Vulnerable** under sub-criteria D1 and D2b and **Data Deficient** under sub-criterion D3.

Identification of biotic indicator

We examined the relevance and data availability/quality for one biotic indicator, selected based on our conceptual model, to assess the risk of collapse from disruption of biotic processes and/or interactions:

- SIGNALT macroinvertebrate assemblage condition score

Indicator: SIGNALT score

Relevance to ecosystem function

While the macroinvertebrate fauna of *Alpine-subalpine Streams* tends to be poorly described, there are numerous species, including stoneflies, mayflies, caddisflies and others that are likely to be restricted to high mountain streams (Campbell et al. 1986). Furthermore, aquatic habitats in Tasmania, along with other alpine regions, are likely subject to summit traps where their isolated populations and limited dispersal abilities prevent migration to other suitable habitat (Boulton et al. 2014), particularly given common adaptations in alpine environments involve the reduction or loss of wings (Campbell et al. 1986). Thus, macroinvertebrate assemblages provide a reliable indicator of biotic function.

Given the paucity of detailed distribution or ecological knowledge, measuring overall condition of the macroinvertebrate assemblage is a reasonable method of assessing biological ecosystem function. SIGNAL 2 is an established method for assessing overall condition of Australian aquatic macroinvertebrate assemblages (Chessman 2003). It involves scoring of genera or families of invertebrates based on their known tolerances of pollution and dissolved oxygen levels (Chessman 2003). SIGNAL 2 has been modified for use by citizen scientists to form the SIGNALT index (National Waterbug Blitz Team 2025).

Data availability and quality

Recent change (D1)

SIGNALT scores from macroinvertebrate assemblage sampling data based on a combination of sampling by the Environment Protection Agency and community groups in VIC, NSW, ACT and TAS were downloaded from the Waterbug Data Portal on the National Waterbug Blitz website (National Waterbug Blitz Team 2025). We identified those sampling locations that fell within a 5-km buffer zone of the gauges selected in Criterion C, to ensure that we selected the most relevant. As for Criterion C, we used two sets – those that fell above the elevation threshold for the alpine/subalpine region in each state (19 sampling locations) and a second larger set associate with the inclusion of additional gauges at lower elevations to ensure that at least 5 gauges were available per state (69 sampling locations). We assumed that these sampling locations were representative of the condition of macroinvertebrate assemblages in *Alpine-subalpine Streams* across the four states, although ACT did not have sufficient sampling locations to analyse separately.

Ten of the 13 alpine/subalpine gauges had records that extended for at least 50 years. For ease of analysis, all years of record were included for all 13 gauges.

Data quality may be variable given the mix of sampling by EPA staff compared with citizen scientists. However, the Waterbug Blitz is an established citizen science program that includes regular high-quality training and quality assurance processes for all data collected. Thus, we are confident that the overall level of data quality is sufficient to support the findings reported herein.

Future change (D2)

We extrapolated any trend in SIGNALT scores to estimate the trajectory of decline for *Alpine-subalpine Streams*, we investigated the relative condition of macroinvertebrate assemblages at sampling locations that fell within 5 km of one of the gauges identified in Criterion C. As for Criterion C, to maximise the available data, we used two sets including those that fell within the alpine/subalpine region (19 sampling locations) but also those that were within 5 km of one of the larger set of gauges, selected to ensure that there were at least 5 per state (69 sampling locations).

Historical change (D3)

Data were unavailable to assess historical change in the condition of macroinvertebrate assemblages.

Selection of collapse threshold

Existing assessments of the relative health of macroinvertebrate assemblages include thresholds for poor condition based on a SIGNALT score. These vary among states: 5.1 in Victoria, 4.7 in Tasmania, 2.9 in NSW and 2.8 in ACT. To set a collapse threshold, we selected a value that was two thirds the value at which the assemblage was poor, representing a substantial loss of condition beyond a point where the assemblage was poor. Thus, the collapse thresholds for SIGNALT score for each state were 3.4 in Victoria, 3.1 in Tasmania and 1.9 in each of NSW and ACT.

Calculation of initial and present/future values

Recent change (D1) and Future change (D2)

SIGNALT score was analysed for each alpine/subalpine sampling location in the time series using the R software environment. We fit a generalised linear model to the data, assessing SIGNALT score as a function of year, elevation and state, including any interaction between elevation and state. This was run for the 19 sampling locations that were in the alpine/subalpine region, as well as the 69 sites in the broader region (Figure 106).

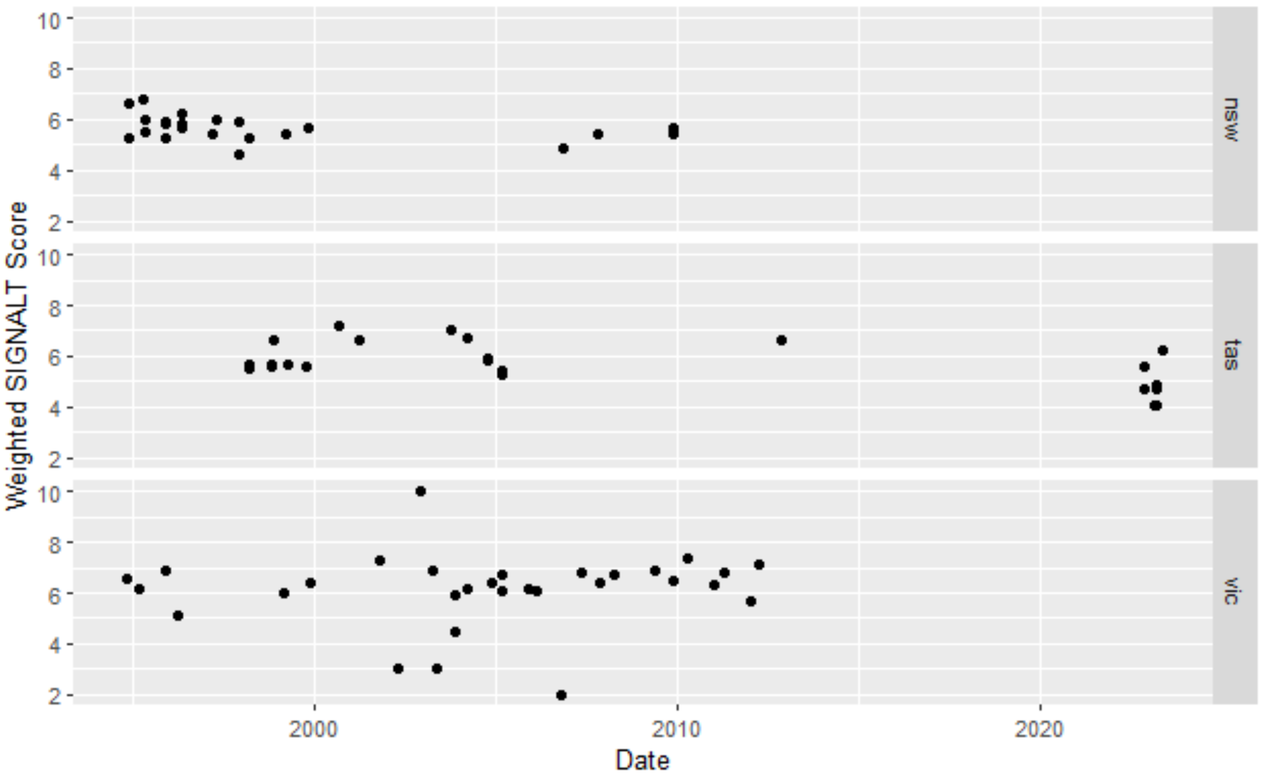


Figure 106. Observed SIGNALT scores over the historical record for 19 sampling locations in the alpine/subalpine zone across three states. Analyses were conducted on the available length of record for each sampling location (range 1994-2023).

Historical change (D3)

Data were unavailable to assess this sub-criterion.

Calculation of severity and extent

Recent change (D1) and Future change (D2b)

There was a highly statistically significant decline in SIGNALT score through time ($F_{1,214} = 8.63$, $p = 0.004$). There were also significant differences in SIGNALT score among states ($F_{2,2} = 7.87$, $p > 0.001$; Figure 107). These findings persisted when additional sampling sites below the relevant elevation thresholds in each state were included, with a statistically significant decline in SIGNALT score through time and differences among states evident. Here, however, there were also significant differences in SIGNAL score across elevations.

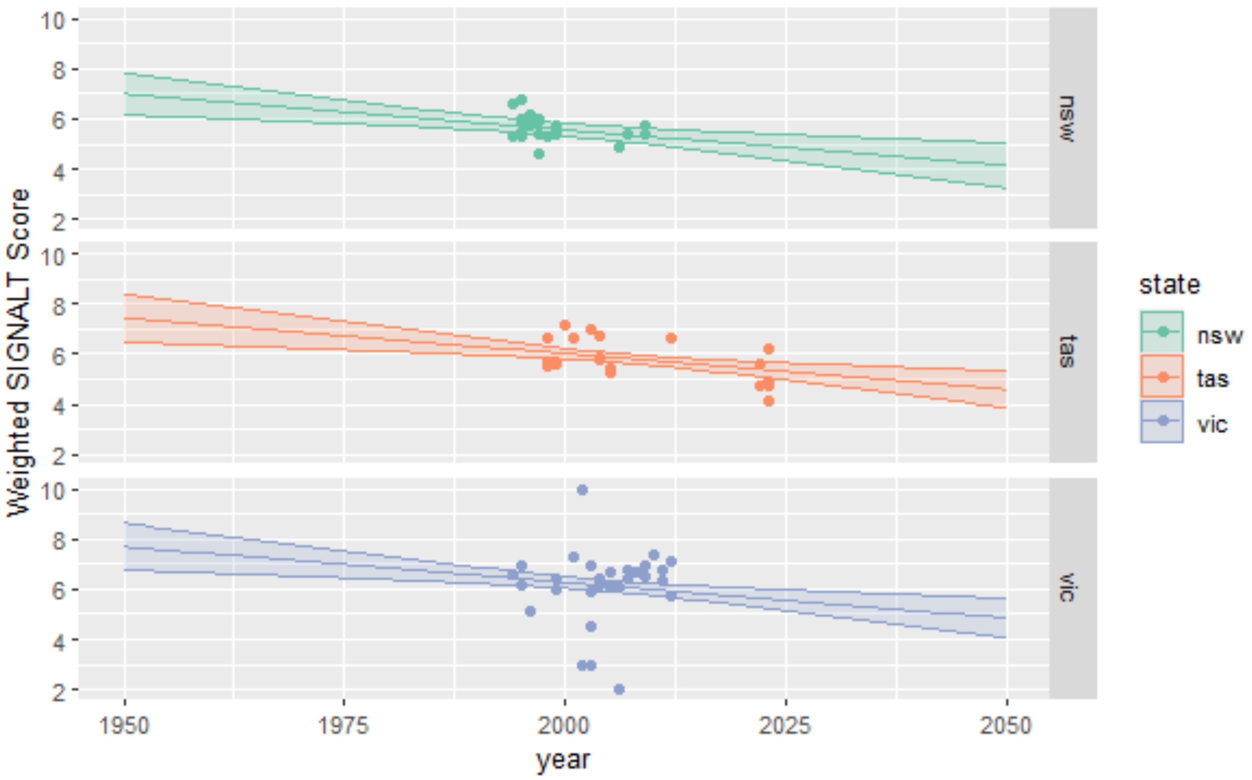


Figure 107. Trend in SIGNALT score extrapolated for the period of 1950 to 2050, including the observed SIGNALT scores over the historical record for 19 sampling locations in the alpine/subalpine zone across three states. Analyses were conducted on the available length of record for each sampling location.

For the period 1970-2020 (relative to sub-criterion D1), SIGNALT score had changed from an estimated 6.44 to 5.01 in NSW, 6.89 to 5.46 in Tasmania and 7.14 to 5.71 in Victoria. This equates to a relative severity of 100 × (6.44 – 5.01)/(6.44 – 1.9) = 31.5% in NSW, 100 × (6.89 – 5.46)/(6.89 – 3.1) = 37.7% in Tasmania, and 100 × (7.14 – 5.71)/(7.14 – 3.4) = 38.2% in Victoria. The risk status under sub-criteria D1 is **Vulnerable.**

For the period 2000-2050 (relative to sub-criterion D2b), SIGNALT score was predicted to change from 5.58 to 4.15 in NSW, 6.03 to 4.60 in Tasmania and 6.28 to 4.85 in Victoria. This equates to a relative severity of 100 × (5.58 – 4.15)/(5.58 – 1.9) = 38.9% in NSW, 100 × (6.03 – 4.60)/(5.58 – 3.1) = 57.7% in Tasmania and 100 × (6.28 – 4.85)/ (5.58 – 3.4) = 65.6% in Victoria. Thus, the risk status under sub-criteria D2b is **Vulnerable**,

Historical change (D3)

There were insufficient data to assess historical changes in SIGNALT score. The risk status is **Data Deficient** under sub-criterion D3.

Criterion E: Quantitative analysis of probability of collapse

Summary

Criterion E requires a simulation model capturing the key features and processes of *Alpine-subalpine Streams* to estimate the probability of collapse with the next 50 to 100 years based on likely future threat scenarios. Here, we utilised existing models of changes in rainfall and runoff associated with 42 CMIP 5 and 37 CMIP6 global climate models (GCMs) for Australia (Zheng et al. 2024) to scale hydrologic gauge data to provide estimates for flows in *Alpine-subalpine Streams* in the future. These scenarios represent possible future climate change by 2046-2075 under the high SSP5-8.5/RCP8.5 global warming scenario (Zheng et al. 2024). Under CMIP6, the median warming was 2.3°C by 2060 (Zheng et al. 2024). Under the median scenarios for each gauge, the average and maximum cease-to-flow durations increase as do the average and maximum duration of low-flow spells. Thus, under this criterion, the risk status is **Endangered** (>20% likelihood of collapse in 50 years).

Identification of abiotic indicators

We examined the relevance and data availability/quality for the same two abiotic indicators used for Criterion C to assess the risk of collapse from disruption of abiotic processes and/or interactions:

- Length of cease-to-flow periods: a measure of the length of periods of time where streams cease to have flowing water in days, including both the median and maximum length of cease-to-flows.
- Length of low flow spells: the duration of time where streams have flows less than the 10th percentile based on the historical time period, measured in days, including both the median and maximum length of low-flow spells.

Indicator: Length of cease-to-flow periods

Relevance to ecosystem function

As for Criterion C, *Alpine-subalpine Streams* are defined as being permanently flowing systems and departure from this constitutes a fundamental change in the character of the ecosystem because many of the biota characteristic of *Alpine-subalpine Streams* cannot tolerate warming or desiccation (Boulton et al. 2014). Thus, the median and maximum lengths of cease-to-flow periods provide a clear indicator of the continued status of *Alpine-subalpine Streams* as near permanently flowing systems.

Model availability and quality

To assess plausible change in the length of cease-to-flow periods under climate change, we used the outputs from 37 GCMs associate with CMIP5 and CMIP6 capturing daily runoff projections for each of the 13 gauges within the alpine/subalpine zones in NSW, Tasmania and Victoria. The projections were provided by Zheng et al. (2024) The gauge data were sourced from BOM as described above. Again, we assume that these gauges are representative of all alpine/subalpine zones, and we again repeated analyses including the highest gauges below the specified elevation thresholds in each state to ensure that there were at least five gauges in each state (for 25 gauges). The results and interpretation were similar for the analyses including all 25 gauges (i.e. including some that were below the elevation threshold for alpine/subalpine regions in some states) compared to those undertaken using the 13 gauges that were within the alpine/subalpine regions and so the analyses using the 25 gauges are not presented here.

We used the daily sequence of projected runoff for each gauge from Zheng et al. (2024) for each of 37 GCMs. We then applied quartile-quartile scaling to the modelled historical flow sequence (also derived from Zheng et al. (2024). This resulted in daily flows for each gauge that had been scaled by the change in daily runoff for the equivalent quartile in the data. For very low flows, quartile-quartile scaling is less effective, so we applied a linear regression in the first quartile. This produced 37 daily sequences of projected flow for each gauge corresponding to 37 projections under plausible future climates.

As noted in Criterion C, data quality was variable, and some gauges had quality codes indicating poor quality data (i.e. >150). These gauges were excluded, leaving 11 gauges in the alpine-subalpine region and 23 in the larger data set.

Selection of collapse threshold

We determined that a median length of cease-to-flow conditions should not exceed 30 days, with a maximum length of less than 90 days within the time series to ensure that biota was not exposed to prolonged periods of desiccation. The 37 GCMs represent a range of possible future outcomes. We use the median projection among those 37 as an estimate of the most likely future climate, and the 90th percentile projection as an estimate of severe climate change. Thus, we define collapse as when the median of 30 days (or maximum of 90 days) is exceeded in the median future climate projection but also noting where those thresholds are crossed in the 90th percentile projection as an assessment of vulnerability to either faster warming than expected, or in the longer term (i.e. beyond 2060).

Calculation of initial and present/future values

Refer to Criterion C for present values.

Calculation of probability of collapse

Under median climate conditions, there was a trend of increasing length of cease-to-flow conditions under future climates (Figure 108). Under the median future climate projection, six of the 11 gauges (excluding 2 outliers) experienced longer cease-to-flow periods and one additional gauge experienced cease-to-flow for the first time (i.e., there was no cease-to-flow for that gauge in the historical record). However, under median future climate change, median length of cease-to-flow did not exceed 30 days for any gauge. Similarly, under the 90th percentile climate projection, median length of cease-to-flow was still longer, but did not exceed 30 days for any gauge.

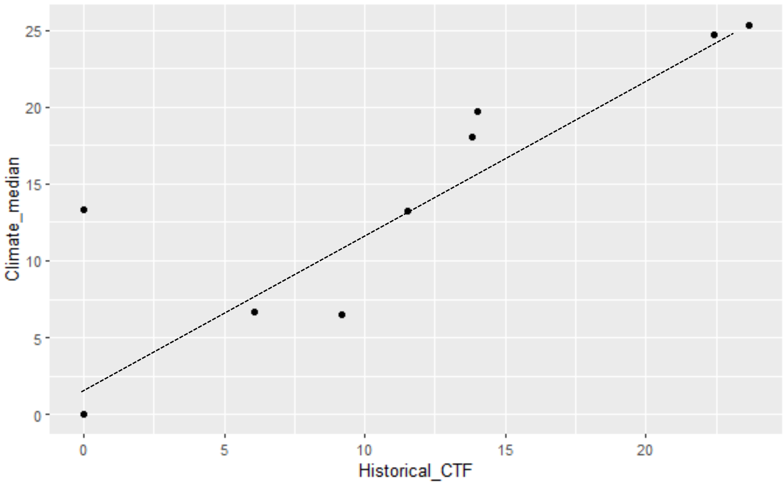


Figure 108. Comparison of median length of cease-to-flow (CTF) (in days) between the historical record (x axis) and the median simulated future climate projection. Each point represents a single gauge. The line illustrates the 1:1 line which indicates no change between the historical and climate change hydrometrics. Points that fall below the line have decreased hydrometric values under median future climate change. Points that fall above the line have increased hydrometric values under median future climate change. For length of cease-to-flow, higher values are a deterioration in condition.

Maximum length of cease-to-flow captured the maximum duration of cease-to-flow conditions across the time series analysed. Under median climate conditions, maximum length of cease-to-flow was at least as long as under historical conditions at all gauges (Figure 109). Under the median future climate projection, seven of the 11 gauges experienced longer cease-to-flow periods and one additional gauge experienced cease-to-flow for the first time (i.e. there was no cease-to-flow for that gauge in the historical record). For those median future climate change conditions, maximum length of cease-to-flow exceeded 90 days for six gauges, suggesting that half of the gauges measured would indicate collapse. Similarly, under the 90th percentile climate projection, maximum length of cease-to-flow also exceeded 90 days for six gauges.

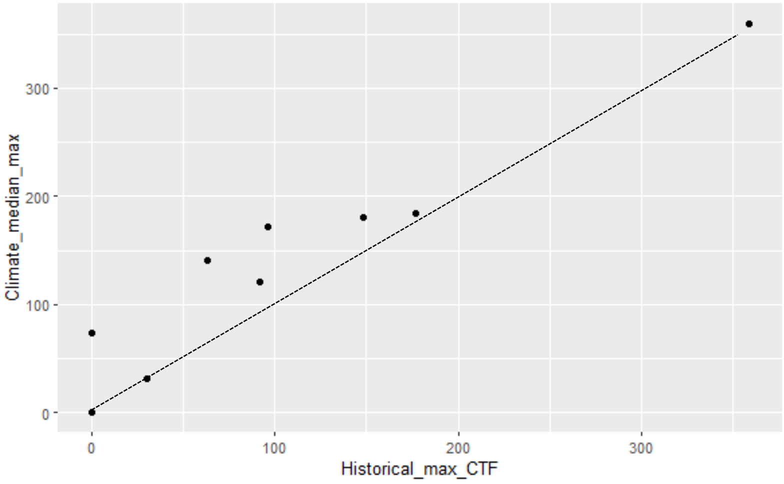


Figure 109. Comparison of maximum length of cease-to-flow (CTF) (in days) between the historical record and the median among the simulated future climate projections. Each point represents a single gauge. The line illustrates the 1:1 line which indicates no change between the historical and climate change hydrometrics. Points that fall below the line have decreased hydrometric values under median future climate change. Points that fall above the line have increased hydrometric values under median future climate change. For length of cease-to-flow, higher values are a deterioration in condition.

Both average and maximum length of cease-to-flows are relevant for the likelihood of collapse, with longer average cease-to-flow periods potentially being more impactful than one-off long cease-to-flows which may be better represented by the maximum. The analysis presented here suggests that average cease-to-flow would not indicate collapse, but that maximum cease-to-flow would indicate collapse at 50% of gauges. These projections are relevant for 2046-2075 depending on the rate at which warming occurs. Thus, these projected changes may occur close to 2046 at a fast rate of warming or closer to 2075 under a slower rate of warming. Given the difference between the two indicators and the uncertainty regarding the rate of warming, we opted to take a conservative assessment of this equating to at least a 20% probability of collapse over 50 years and therefore a risk status of **Endangered** under this indicator.

Indicator: Length of low-flow spells

Relevance to ecosystem function

As noted above, *Alpine-subalpine Streams* are defined as being permanently flowing systems and departure from this constitutes a fundamental change in the character of the ecosystem. The median and maximum lengths of low-flow spells provide a second line of evidence regarding the continued status of *Alpine-subalpine Streams* as near permanently flowing systems. This second line of evidence is prudent given the challenges associated with the measurement and modelling of cease-to-flow periods, which can be difficult to adequately capture and model.

Data and model availability and quality

As for the length of cease-to-flow periods hydrologic indicator, we again used the outputs from 37 GCMs associate with CMIP5 and CMIP6 capturing daily runoff projections for each of the 11 gauges within the alpine/subalpine zones in NSW, Tasmania and Victoria. Again, we assume that these are representative of all alpine/subalpine zones, and we again repeated analyses including the highest gauges below the specified elevation thresholds in each state to ensure that there were at least five gauges in each state (for 23 gauges). The results and interpretation were again similar for the analyses including all 23 gauges (i.e. including some that were below the elevation threshold for alpine/subalpine regions in some states) and so those analyses are not presented separately here.

As previously, we used the daily sequence of projected runoff for each gauge from Zheng et al. (2024) for each of 37 GCMs. We again used quartile-quartile scaling to produce scaled daily flows for each gauge. This produced 37 daily sequences of projected flow for each gauge corresponding to 37 projections under plausible future climates.

Selection of collapse threshold

We determined that a median duration for a low-flow spell should not exceed 60 days, with a maximum duration of less than 120 days within the time series to ensure that biota was not exposed to prolonged periods of desiccation. The 37 GCMs again represent a range of possible future outcomes. As for length of cease-to-flow, we use the median future climate projection as an estimate of the most likely future climate, and the 90th percentile climate projection as an estimate of severe climate change. Thus, we define collapse as when the median of 60 days (or maximum of 120 days) is exceeded in the median future climate projection, but also noting where those thresholds are crossed in the 90th percentile projection as an assessment of vulnerability to either faster warming than expected, or in the longer term (i.e. beyond 2060).

Calculation of initial and present/future values

Refer to Criterion C for present values.

Calculation of probability of collapse

Under median climate conditions, there was very little change in the median length of low-flow spells for gauges that had historically experienced low flows (Figure 110). However, four gauges that had not historically experienced low-flow spells under historical conditions were projected to have low flows (out of the 11 gauges analysed). Despite this increase in the number of gauges experiencing low-flow spells, under median future climate change, median length of cease-to-flow did not exceed 60 days for any gauge. Similarly, under the 90th percentile climate projection, the median length of low-flow spells did not exceed 60 days for any gauge.

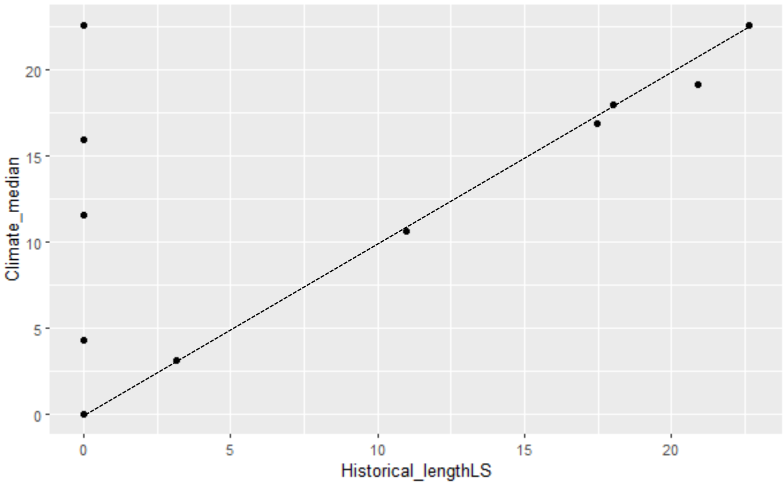


Figure 110. Comparison of median length of low-flow spells (in days) between the historical record and the median simulated future climate projection. Each point represents a single gauge. The line illustrates the 1:1 line which indicates no change between the historical and climate change hydrometrics. Points that fall below the line have decreased hydrometric values under median future climate change. Points that fall above the line have increased hydrometric values under median future climate change. For length of cease-to-flow, higher values are a deterioration in condition.

Maximum length of low-flow spells captured the maximum duration of low-flow conditions across the time series analysed. Under median climate conditions, maximum length of cease-to-flow was again at least as long as under historical conditions at all gauges (Figure 111). The same four additional gauges recorded low-flow conditions as was observed for the median length of low-flow spells indicator. Under the median future climate projection, two of the 13 gauges experienced low-flow spells longer than 120 days, indicating collapse for those gauges. The same pattern was apparent under the 90th percentile climate projection. This suggests there is a low probability (~15%) of collapsing in the next 50 years, and therefore the risk status if Least Concern under this indicator.

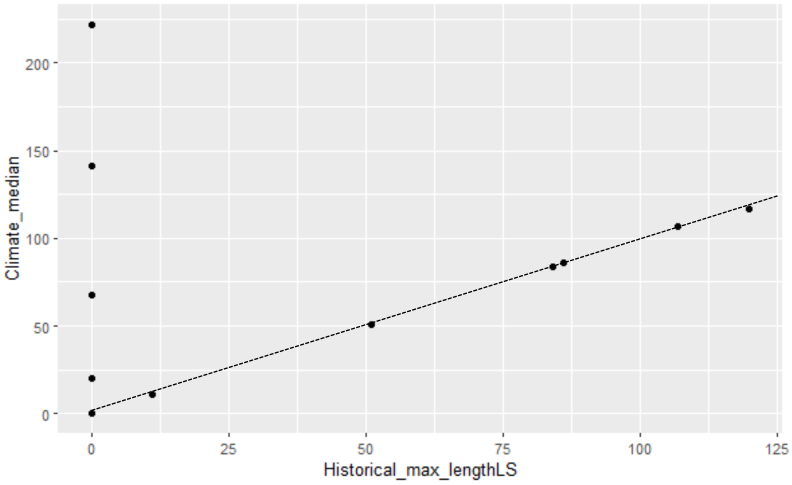


Figure 111. Comparison of median length of low-flow spells (in days) between the historical record and the median among the simulated future climate projections. Each point represents a single gauge. The line illustrates the 1:1 line which indicates no change between the historical and climate change hydrometrics. Points that fall below the line have decreased hydrometric values under median future climate change. Points that fall above the line have increased hydrometric values under median future climate change. For length of cease-to-flow, higher values are a deterioration in condition.

Maximum length of low-flow spells captured the maximum duration of low-flow conditions across the time series analysed. Under median climate conditions, maximum length of cease-to-flow was again at least as long as under historical conditions at all gauges (Figure 112). The same four additional gauges recorded low-flow conditions as was observed for the median length of low-flow spells indicator. Under the median future climate projection, two of the 11 gauges experienced low-flow spells longer than 120 days, indicating collapse for those gauges. The same pattern was apparent under the 90th percentile climate projection. This suggests there is a low probability (~15%) of collapsing in the next 50 years, and therefore the risk status if **Least Concern** under this indicator.

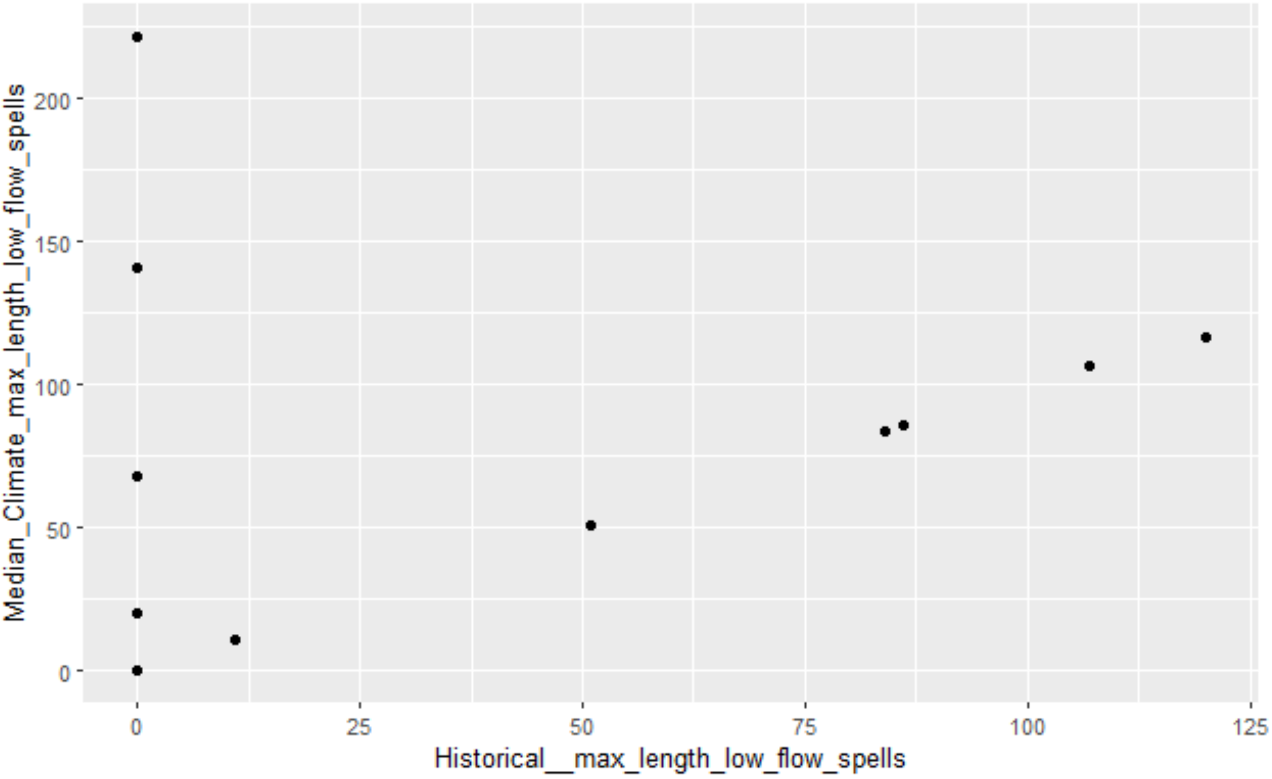
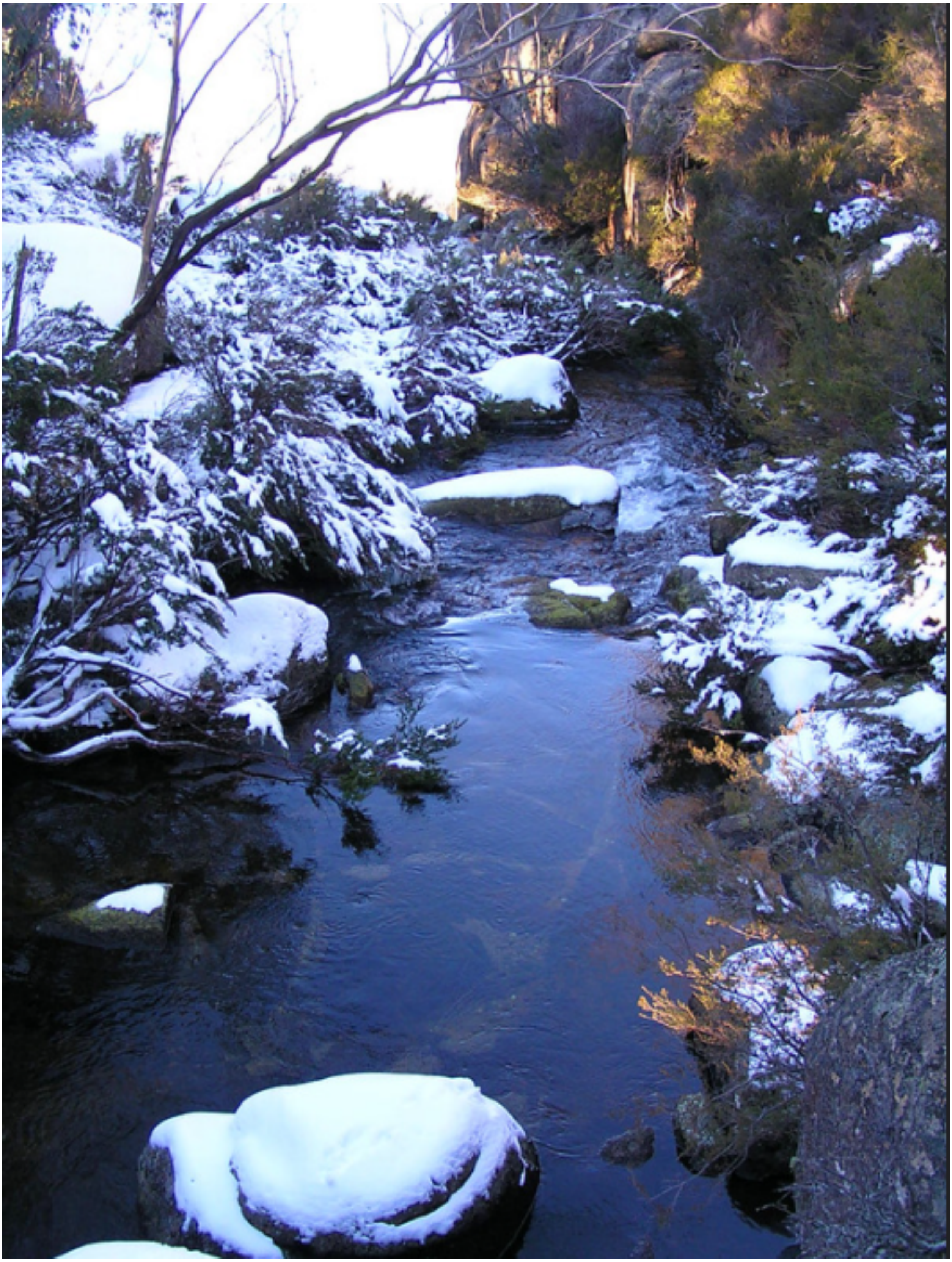


Figure 112. Comparison of maximum length of low-flow spells (in days) between the historical record and the median among the simulated future climate projections. Each point represents a single gauge. The line illustrates the 1:1 line which indicates no change between the historical and climate change hydrometrics. Points that fall below the line have decreased hydrometric values under median future climate change. Points that fall above the line have increased hydrometric values under median future climate change. For length of cease-to-flow, higher values are a deterioration in condition.



Crystal Brook in winter, Mount Buffalo National Park, Australian Alps collection. Photo: Parks Australia.

Alpine-subalpine Lakes

Authors

Rebecca E Lester, Michael Traurig, Galen Holt, Emily Armstrong, Jessica Rowland

Reviewers

Anikó B. Tóth, José R. Ferrer-Paris

Contributors

Arn Tolsma, Ken Green, Jennie Whinam, Emily Nicholson

Biome

F2 Lakes

Functional group

F2.2 Small permanent freshwater lakes; F2.4 Freeze-thaw freshwater lakes

IUCN status

Critically Endangered

Assessment Summary

Alpine-subalpine Lakes is restricted to elevations of 915 m in Tasmania and 1,500 m in New South Wales. It is defined by standing permanent water bodies found within the alpine-subalpine region. The ecosystem is threatened by climate change, invasive species and changes to fire regimes and infrastructure development. The status of the ecosystem is assessed as **Critically Endangered** as assessed under Criterion D2b (Table 40). There were insufficient data to assess the change in the functional features of the ecosystem over time. Lake biota indices indicate that the condition of lake macroinvertebrate assemblages have declined through time, and assessment of the likely future change in ecosystem function suggested that collapse thresholds may be reached by 2050.

Table 40. Summary of the IUCN Red List of Ecosystems assessment of the Australian *Alpine-subalpine Lakes*.

Criteria	**A**	**B**	**C**	**D**	**E**	**Overall**
Sub-criterion 1 (past 50-years)	DD	LC	LC	EN	DD	CR
Sub-criterion 2 (50-year period including present & future)	DD	LC	LC	CR		
Sub-criterion 3 (since ~1750)	DD	LC	DD	DD		

Criteria: A = reduction in distribution; B = restricted distribution; C = environmental degradation; D = disruption of biotic processes; E = quantitate probability analysis. Sub-criteria are only applicable to criteria A-D. Risk categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient. Parentheses indicate plausible bounds. Overall represents the highest risk rating across all assessed sub-criteria

Ecosystem Description

Ecosystem Classification

Alpine-subalpine Lakes aligns with both F2.2 Small permanent freshwater lakes and F2.4 Freeze-thaw freshwater lakes under the IUCN Global Ecosystem Typology (Keith et al. 2022a). Most *Alpine-subalpine Lakes* is not listed under Commonwealth, State or Territory legislation, but one exception is Blue Lake in the Kosciuszko National Park, New South Wales. Blue Lake is a wetland of international importance under the Ramsar Convention on Wetlands (DAWE 2020; DPIE 2020). Blue Lake is classified as a high-altitude marsh on inorganic/peat soils ("Va"; see DECC 2008).

Alpine-subalpine Lakes in Australia is classified according to the extent to which lakes mix (see Boulton et al. 2014). Under this classification, *Alpine-subalpine Lakes* are cold monomictic (mix once a year, freeze over in winter but mix in summer when they warm and so do not maintain stratification), dimictic (mix twice per year during spring and autumn after stratification under ice cover and in summer due to thermal gradients) and polymictic (mix frequently, sometimes continuously and stratification is in response to the diel temperature changes, most common in very shallow lakes) (Bayly & Williams 1973; Tyler 1974, 1992; Green 2011, 2012; Boulton et al. 2014). Some deep lakes formed by the action of glaciers in the last ice age (glacial lakes) in Tasmania are also meromictic (mix incompletely and infrequently) and these are otherwise uncommon in Australia.

Distinction from similar ecosystem types

Alpine-subalpine Lakes occur in mountainous regions, typically at elevations above the tree line (Čiamporová-Zat'ovičová 2011). *Alpine-subalpine Lakes* are often groundwater dependent and can be fed by snowmelt (Boulton et al. 2014). Unlike non-alpine lakes, *Alpine-subalpine Lakes* can freeze in winter. Alpine wetlands such as *Alpine Sphagnum Bog and Associated Fen* are waterlogged areas which, like alpine lakes, are fed by groundwater, precipitation and snowmelt, however they differ in size (with lakes typically larger; e.g., >8 ha; Butt et al. 2021), lack of trees, shrubs and other vegetation within the waterbody (excluding macrophytes, which may be present) and geomorphology (where lakes are found in topographic depressions; Butt et al. 2021). Lakes can have more permanent water and are frequently deeper (Butt et al. 2021). Alpine-subalpine Lakes are considered excellent sentinels for entire mountain ecosystems, providing early warning of environmental changes such as atmospheric pollution and climate change (Čiamporová-Zat'ovičová 2011).

Distribution

Both natural and non-natural lakes occur in the alpine-subalpine regions of Australia (Figure 113). For the purposes of this assessment, only permanent, natural waterbodies (Keith et al. 2020) > 1 ha in area are included. Natural *Alpine-subalpine Lakes* are located between 145.58° to 148.32° longitude and -36.40° to -43.50° latitude (Figure 113), and occur above 1500 m in New South Wales, and 915 m in Tasmania (Costin 1957).

Australian *Alpine-subalpine Lakes* cover an area of approximately 99 km^2 (Figure 113). Spatial products used to create this map represent the most accurate, national mapping available at the time of assessment.

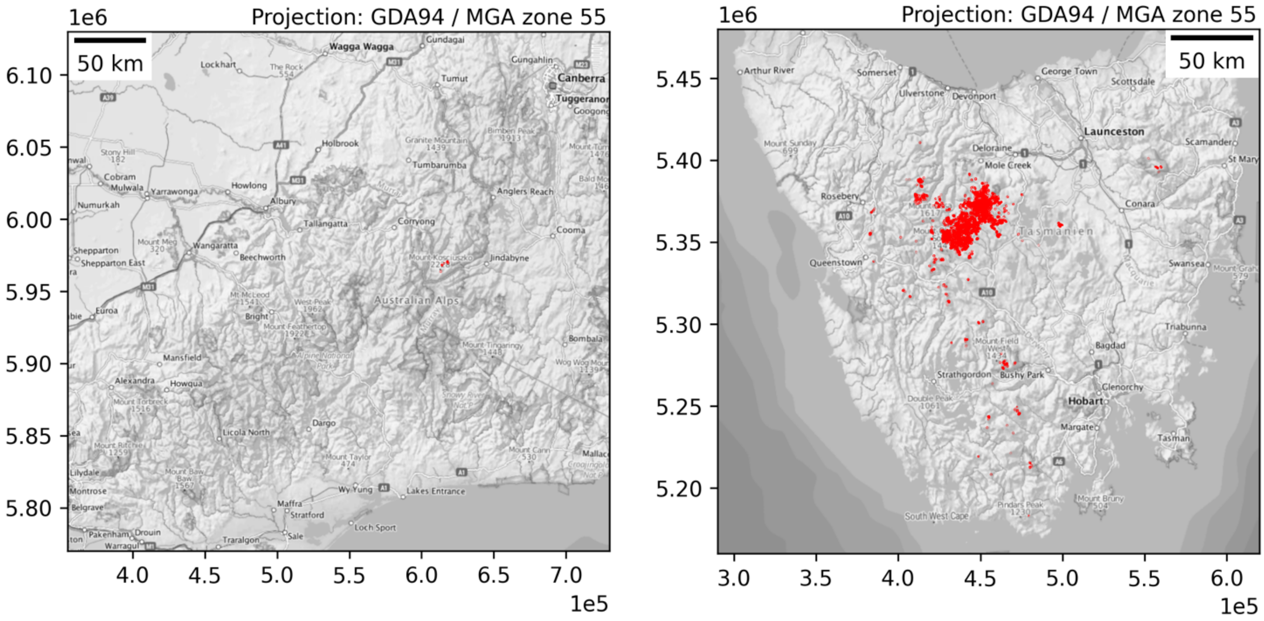


Figure 113. Distribution of natural *Alpine-subalpine Lakes* (red) occurring above 915 m elevation in Tasmania (right) and above 1500 m elevation on mainland Australia (i.e. New South Wales) (left).

Abiotic environment

Mainland Australian alpine and sub-alpine lakes occur between 1500 m and 2070 m above sea level. There are five alpine lakes in the Australian mainland all of which are in Kosciuszko National Park, NSW: Cootapatamba, Albina, Club, Blue, and Hedley Tarn. The maximum depth of the lakes ranges from 1.6 to 29 m, and they are between 1.5 to 14.4 ha in size (Green 2011). The lakes are mainly cold monomictic, but Blue Lake is dimictic (Green 2011, 2012). Outflow water temperatures typically remain below 1 °C when the lake surface is frozen, but outflow water temperatures may rise to 20 °C by mid-to-late summer, (Figure 114; Green 2011). The lakes have low total dissolved solid concentrations (e.g., Blue Lake 2.5 mgL^{-1}; Williams et al. 1970; Tyler 1974; Buckney 1980) and the ionic proportions are $Na^{+} > Ca^{2+} >< Mg^{2+} > K^{+}$: $SO_4^{2-} > Cl^{-} > HCO_3^{-}$.

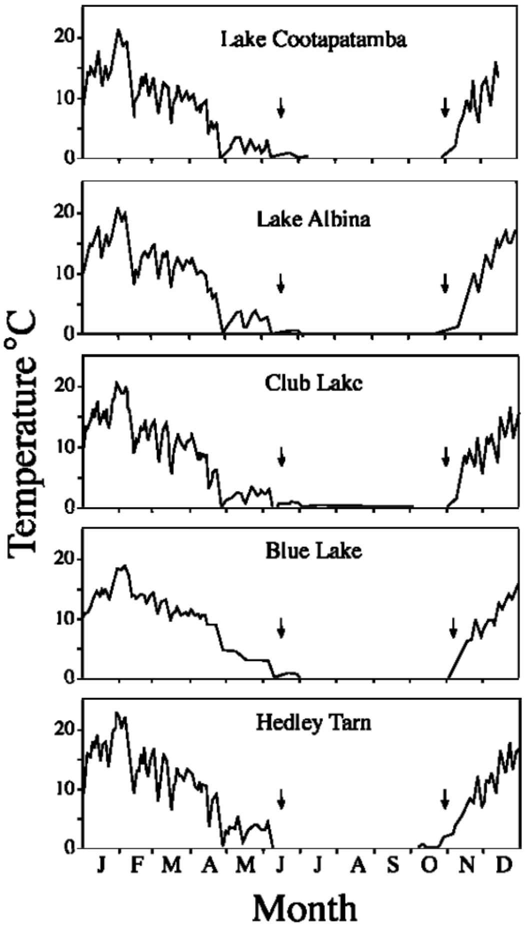


Figure 114. Comparisons of the seasonal mean daily temperature changes in outflow of the five lakes for 2009. Arrows mark approximate ice-on and ice-off dates. Figure from (Green 2011).

Tasmanian alpine and sub-alpine lakes typically occur between 650 m and 1100 m above sea level (Tyler 1974). There are around 1300 lakes >1 ha, which cover the Tasmanian Central Plateau, the mountains of the Southwest, and the Ben Lomond massif. *Alpine-subalpine Lakes* vary in depth from < 10 m to 166.5 m (Tyler 1992) and are between < 1 to 150 km^2 in size (Tyler 1974). Surface water temperatures of these lakes generally range from 2 to 19 °C, and do not regularly freeze (as on the mainland) (Tyler 1974). Lakes on the eastern central plateau are exposed, outside of direct glacial activity during the last ice age, do not stratify (i.e. are thermally homogenous) and are classed as cold polymictic (Tyler 1974). However, the western edge of the plateau has glacial lakes which are deep and sheltered (e.g., Lake St Clair), and that possibly stratify in summer (Tyler

1974). It is also possible that some of the deep, sheltered lakes in Tasmania may be dimictic (Tyler 1974).

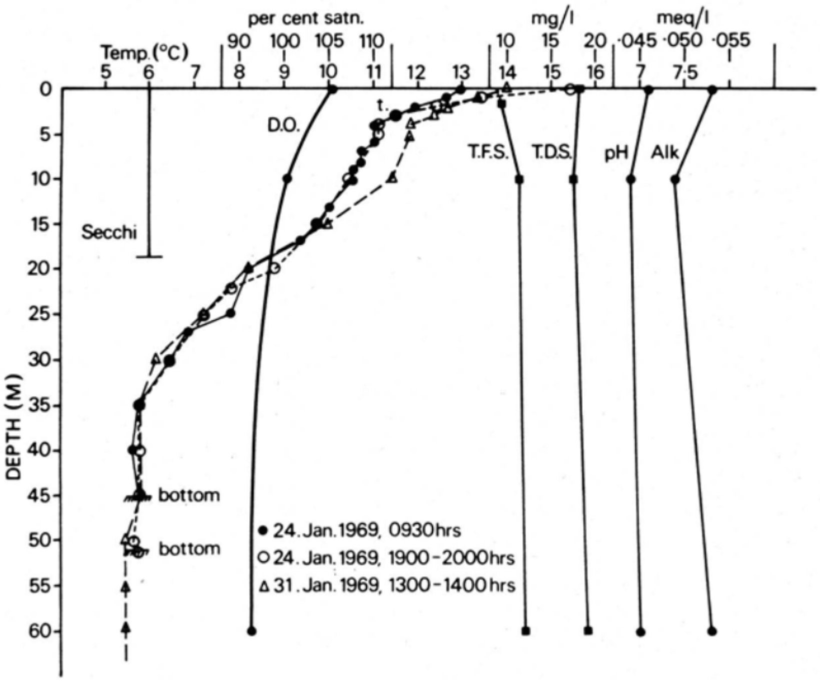


Figure 115. Temperature-depth profile and other parameters of a warm monomictic lake (Lake Picton, 838 m asl) in south-western Tasmania. Figure from Tyler (1974).

In Tasmania, there is a clear difference in geological and climatic features between East and West, which influence the water chemistry of the *Alpine-subalpine Lakes.* The western mountains have a Precambrian, Cambrian and Ordovician origin with siliceous and peaty soils (Vanhoutte et al. 2006), whereas the eastern province is dominated by Jurassic dolerite and rich acidic soils from the igneous rocks. The western area has high rainfall which declines towards the east (Tyler 1992). The lakes in Southwestern Tasmania (on the Precambrian quartzites) tend towards World Average Salt Water in the ionic proportions, due to the marine influence with precipitation (Williams 1964; Buckney & Tyler 1973). The ionic proportions are $Na^+ > Ca^{2+} >< Mg^{2+} > K^+$: $Cl^- > HCO_3^- > SO_4^{2-}$. The total dissolved solids (TDS) are low (<20 mgL^{-1}) (Figure 115; (Buckney & Tyler 1973; Buckney 1980). As rainfall declines to the East, the lakes display enrichment of Ca^{2+} and HCO_3^- (Knott et al. 1978; Hart & McKelvie 1985) attributed to weathering of the bedrock. These lakes are also low in TDS (<50 mgL^{-1}) and the ionic proportions are closer to World Average Fresh Water; $Ca^{2+} > Na^+ >< Mg^{2+} > K^+$: $HCO_3^- > Cl^- > SO_4^{2-}$.

In general, alpine lakes have a set of characteristics in common. They tend to be relatively small, with low water temperature (usually below 12°C with a maximum of 15°C during warm years), with very low light regimes but high UV radiation and high transparency (Čiamporová-Zat’ovičová 2011). Most alpine lakes have few available nutrients and so are clear, oligotrophic, dilute and relatively unproductive compared to other lake ecosystems (Čiamporová-Zat’ovičová 2011). Spring snowmelt is a dominant hydrologic event (Čiamporová-Zat’ovičová 2011).

Characteristic native biota

Australian *Alpine-subalpine Lakes* contain a range of endemic fish and invertebrate species. Biomass is generally low, for example, 1.2 to 5.9 g m^{-2} in Lake Cootapatamba and 3.9 to 19.5 g m^{-2} in Lake Albina (Hancock et al. 2000; Timms 2002; Timms et al. 2013). Only a few lakes have had the benthos surveyed (Timms 1980). Mainland alpine lakes in the Kosciuszko National Park have lower diversity (~10 benthos species) than the Tasmanian lakes (~17 benthos species; (Timms 1980; Timms et al. 2013). Larger Tasmanian lakes typically have 20-25 benthos species, dominated by the Oligochaeta (worms), Chironomidae (Midges), Amphipoda, and Bivalvia (pea shells) (Leonard & Timms 1974). Smaller Tasmanian lakes have an average of <15 species, made up of Chironomidae, Phreatoicidae (Freshwater isopods) and Gammaridae (Amphipods) (Leonard & Timms 1974; Knott et al. 1978; Timms 1980).

In the upper layers of the lake strata, crustaceans dominate the biomass, if introduced trout are absent (Knott et al. 1978). The syncarid crustacean (*Anaspides* spp.) are common, including *Anaspides swaini* (Figure 116), *A. jarmani, A. richardsoni* and *A. spinulae* (Ahyong 2016). *Ameletoides lacusalbinae* (Nesameletidae) has only been described from Lake Albina in the Kosciuszko National Park (Suter & Webb 2012), and the alpine crayfish (*Euastacus reiki*) occurs in Blue Lake in Kosciuszko National Park (Lawler & Crandall 1998). Lake Cootapatamba had a zooplankton fauna characterised by few taxa, including *Boeckella montana*, *B. pseudochelae*, *Daphnia nivalis* and the rotifer *Keratella slacki* was also dominant but was temporally variable (Benzie 1984).

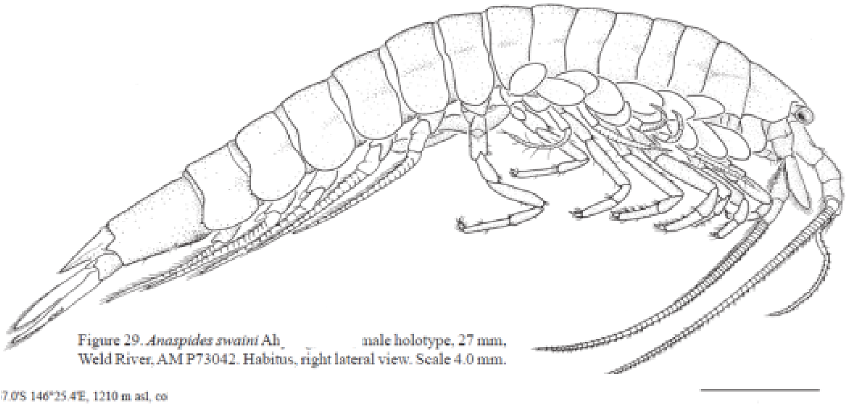


Figure 116. *Anaspides swaini*. Figure from Ahyong (2016).

Lake insect fauna is dominated by Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies; Dean & Cartwright 1992). Stoneflies are considered an important faunal group, with high divergence of species found in alpine areas. Three alpine species in the genus Riekoperla have been identified using molecular taxonomy and other endemic stonefly and mayfly species are likely present but undescribed (Williams et al. 2014). Many aquatic insects use fringing lake vegetation as habitat and emergence sites. This vegetation comprises diverse communities (including herbfield, heath, fens/bogs; see Costin & Wimbush 1972) of varying condition. If the fringing vegetation is threatened (e.g., trampling by introduced horses or deer) then there is potential for aquatic insects to subsequently be affected.

Characteristic fish to mainland alpine lakes are typically high-elevation *Galaxias* species, including *Galaxias supremus* sp. nov. (Figure 117; Raadik 2014), *G. findlayi* (Good 1992) and *G. olidus* (Green 2008), In Tasmanian Alpine lakes, characteristic species include Western paragalaxias (*Paragalaxias julianus*), Great Lake paragalaxias (*P. eleotrodies*), Shannon paragalaxias (*P. dissimilis*), Saddled galaxias (*Galaxias tanycephalus*) and Arthurs paragalaxias (*P. mesotes*). The distribution and abundance of fish in Tasmanian Alpine-subalpine Lakes have been altered by introduced Brown Trout (*Salmo trutta*) and Rainbow Trout (*Oncorhynchis mykiss*) through competition with endemic species and predation (Fulton & Tyler 1993; Hardie et al. 2006). Some alpine lakes can be fishless as a result of natural migration barriers (Čiamporová-Zat'ovičová 2011). Lake Cootampatamba was historically fishless (Green 2014). Other vertebrate fauna includes frogs such as the Critically Endangered Baw Baw frog (*Philoria frosti*) and red-crowned toadlet (*Pseudophryne australia)*, both of which have alpine development (Hazell 2003) which can use lakes but are more commonly found associated with other wetland habitats, favouring moist conditions. Platypus (*Ornithorhynchus anatinus*) have also been recorded in alpine-subalpine lake environments in Tasmania (Otley et al. 2000).

Figure 117. *Galaxias supremus* sp. nov. collected from Blue Lake. Figure from (Raadik 2014).

There are few, if any, macrophytic communities in the benthic or littoral zones of the Australian alpine and subalpine Lakes (e.g., Blue Lake has none; DECC 2008). Photosynthesis is largely carried out by phytoplankton such as *Ulothrix*, *Staurastrum*, *Staurodesmus*, *Arthodesmus*, *Zygnema*, *Mougeotia*, *Rhizoclonium* and *Oscillatoria* (DECC 2008), some of which are endemic to *Alpine-subalpine Lakes* such as *Micrasterias hardyi* and *Staurastrum sagittarium* (Tyler 1992). In Tasmania, dinoflagellates are a major component of the phytoplankton and include eight species from the *Peridinium* genus (of which *P. lomnickii* and *P.* sp. 2 are alpine-restricted mixotrophs), *Thecadiniopsis tasmanica* (a coastal to alpine species), and *Prorocentrum playfairi* (a coastal to highland species) (Ling et al. 1989). Zooplankton (including *Asplanchna* spp., *Attheyela* [*Delachauxiella*] spp., *Alona* spp., *Camptocercus* spp., *Eucyclops* spp., *Chydorus* spp., *Boeckella montana* and *Macrothrix* spp.) and rotifers (including *Keratella*, *Polyarthra* and *Monommata*) (Bayly 1970) feed on the phytoplankton. In Tasmania, microfaunal groups (protozoans, rotifers and microcrustaceans) are diverse and comprise many endemic species, however information is sparse (Fulton & Tyler 1993).

Key components, processes, and interactions

Competition and predation dynamics within and between trophic levels influence the diversity and abundance of biota in *Alpine-subalpine Lakes* (Figure 118). Phytoplankton (dominated by green algae including desmids and the *Chlorococcales*) are preyed upon by zooplankton (dominated by the copepod crustaceans *Boekella* and *Calamoecia*) and benthic fauna which, in turn, are consumed by higher order predators such as fish (e.g., *Galaxias*). Fringing littoral vegetation can be a key habitat for invertebrates and fish (Boulton et al. 2014). Changes in taxa can have flow-on effects to other trophic levels. For example, the migration of *Galaxia olidus* into Lake Cootabatamba anecdotally led to the loss of two benthos species from the lake (Green, pers. comm., 2020).

The input of heat and light influences lake thermal characteristics and thus the stratification of Australian alpine and sub-alpine lakes. Reduced penetration of light and heat (e.g., as a result of algae or sediment in the water column) can alter stratification characteristics (Closs et al. 2009). Generally, alpine-subalpine waters are very clear and allow light to penetrate to deeper waters. However, this can depend on surface condition (e.g., waves scatter light, reducing penetration) and the particulate materials and suspended sediments (which also scatter light; Closs et al. 2009). Stratification influences the availability of nutrients and gases – with availability reduced in deeper waters (Closs et al. 2009). In turn, this can limit the diversity and abundance of organisms occurring in deeper waters, such as macrophytes, phytoplankton, zooplankton and benthic macroinvertebrates, and the biota that consume them (Closs et al. 2009). For example, high light penetration in alpine-subalpine lakes provides the opportunity for algal growth (Boulton et al. 2014) within the euphotic zone (up to 30 m depth; (Boulton et al. 2010), which can lead to alterations in water quality and impacts on fish populations through introduced toxins and decreased oxygen levels.

Water depth is an important characteristic that influences the temperature and thus stratification of Australian alpine and sub-alpine lakes. While depth of Australian alpine-subalpine lakes is extremely stable, fluctuations in alpine lake levels may occur due to rainfall and groundwater inputs and snow (which can temporarily increase the volume and depth of the lake; Saber et al. 2020). Glaciated cirque lakes (e.g., Blue Lake) and morainal lakes (e.g., Hartz Lake) are deep and generally sheltered (Benzie 1984), which may enable ice cover or summer stratification. In contrast, shallow exposed lakes may freeze in winter but are unlikely to develop sustained thermal stability (Boulton et al. 2014).

Ice forms on the surface of some Australian alpine and sub-alpine lakes. The formation of surface ice can promote stratification as turbulence and mixing decreases, leading to stronger gradients in temperature, oxygen and nutrients (Adrian et al. 1999). The season in which ice forms on the lake surface has reduced in the last fifty years: in the 1970s, the period of ice cover was June-November/December, but the ice now breaks up earlier, in September/October (Green 2011). The earlier break-up of ice is related to earlier snow thaw which is, in turn, associated with increased spring temperatures and regional warming (Green 2010). Earlier ice break up can considerably change hydrological, chemical, and biological lake properties (as observed in countries such as the USA; Preston et al. 2016). Earlier ice melts may lead to longer periods of direct sunlight increasing surface water temperatures which, in turn, can lead to longer stratified periods in summer. In addition, changed timing of ice melts may have impacts on the biological timing of events such as algal blooms and breeding cycles of aquatic organisms.

In terms of general lake water chemistry (i.e. nutrient and ionic content/gradients), phytoplankton, benthic algae, and mosses increase the concentration of oxygen in the water column via photosynthesis (Boulton et al. 2014). Wind action dissolves atmospheric oxygen in surface waters (and facilitates mixing via turbulence in the water column; Boulton et al. 2014). Conversely, respiration by plants, animals and aerobic bacteria can decrease oxygen concentration and elevate CO_2 concentration (Boulton et al. 2014). However, it is unlikely that *Alpine-subalpine Lakes* will show great variation in concentrations of CO_2 or dissolved oxygen, as they are cool and oligotrophic (low in nutrient concentration). Similarly, it is unlikely that *Alpine-subalpine Lakes* show great variation in ionic content unless significant inputs (such as fires) or greatly increased evaporation (arising from reduced precipitation) occurs altering pH, nutrients and salinity. The oliogotrophic nature of many alpine-subalpine lakes in Australia can limit biological components of the ecosystem (e.g., algal growth), as can the depth and stratification type of the lake (Čiamporová-Zat’ovičová 2011).

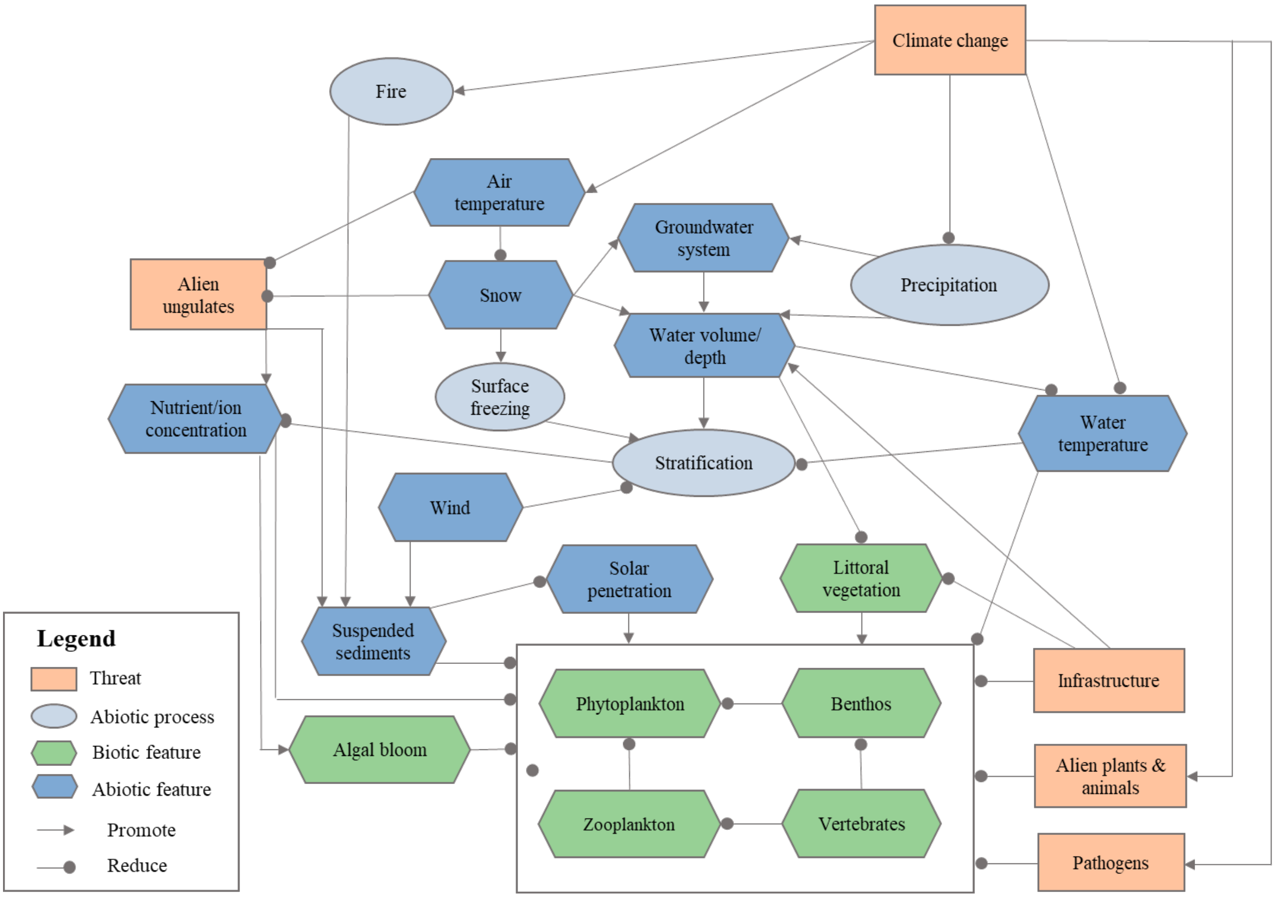


Figure 118. Conceptual diagram illustrating relationships between, and threats to, biotic and abiotic components and processes in *Alpine-subalpine Lakes*

Major threats

Climate change

Climate change is a key threat to Australian *Alpine-subalpine* Lakes (Čiamporová-Zat'ovičová 2011). Climate change can alter precipitation, water temperature and air temperature which, in turn, can influence stratification and nutrient availability in alpine lakes. Minor changes in temperature can alter hydrologic cycles and the extent and duration of ice cover and thermal and light regimes (Čiamporová-Zat'ovičová 2011). These changes then alter the duration of growing seasons, water level fluctuations and the structure and productivity of lake ecosystems (Čiamporová-Zat'ovičová 2011). For example, unseasonal high air temperatures contribute to a reduction in snowpack in nearby mountains. This results in exposure to heavy rain and increased run-off, which in turn can increase sedimentation (Perga et al. 2018). Severe rainfall events that resulted in high turbidity can strongly modify lake hydrodynamics (e.g., by altering stratification and mixing dynamics) and metabolism (Perga et al. 2018). Earlier thawing of ice on the lake surface can also influence stratification (Green 2011), altering the water chemistry and timing and frequency of mixing, fundamentally changing the properties of the lake and specific niche habitats (Closs et al. 2009). Increased temperatures also have the capacity to change algal dynamics (Closs et al. 2009), which in turn, may reduce oxygen availability and pose a threat to native biota (Boulton et al. 2014). Climate change also plays an important role in both altering the life histories of native biota (Boulton et al. 2014), and can facilitate the arrival of invasive species, both terrestrial ungulates and plants and aquatic animals and plants, likely due to increases in air temperature and less snow cover (McDougall & Broome 2007; Alexander et al. 2016). As water temperatures increase, so does the potential for invasion of lower altitude species into alpine areas (e.g., Green 2014). Similarly, raised air temperature enables the invasion of ungulate species.

Invasive species

Invasive species are another important threat to Australian *Alpine-subalpine Lakes* including terrestrial and aquatic biota, pathogens and disease. Novel species assemblages can cause competition for resources or direct mortality through predation, and lead to native species losses (as observed overseas; (MacLennan 2015). For example, the introduction of black worms to Lake Albina via firefighting equipment anecdotally led to the loss of two benthos species (K. Green, pers. comm., 2020). Introduced fish, including (typically non-alpine) native galaxias have been identified as another threat to the lacustrine benthos (Tyler 1974; Knott et al. 1978; Timms 1980; Timms et al. 2013). For example, mountain galaxias (*Galaxias olidus*) invaded the historically fishless Lake Cootapatamba resulting in loss of endemic invertebrate species (Green 2014). On land, invasive ungulates, primarily horses and deer, threaten alpine lakes through trampling, causing sedimentation – and through defecation- and nutrification through defecation (De Deckker et al. 2023). Both of these processes can reduce water quality and thus negatively affect habitat suitability for native biota (Boulton et al. 2014). Introduced pathogens are also a threat to Australian alpine and sub-alpine lakes. There is potential for chytrid fungus, mucor amphibiorum (which affects platypus), and the highly invasive diatom (*Didymosphenia geminata*, colloquially known as rock snot) to be introduced through human or animal vectors (each of which has been responsible for extensive species losses and population reductions elsewhere (Clemann et al. 2009; Gust & Griffiths 2009; Kelly 2009).

Other threats

Finally, other anthropogenic influences, including modified fire regimes and the development of infrastructure have the potential to threaten alpine lakes. Increased fire frequency can alter wetland pH, nitrogen content, and salinity (Kotze 2013), although fire frequency-induced changes have not yet been observed in Australian *Alpine-subalpine Lake* ecosystems. The flora and fauna of Australian alpine systems have, however, frequently been exposed to fire and are considered highly resilient (Williams et al. 2008).

The development of any new dam infrastructure could change hydrological flows, impacting water depth, stratification, and sediment loads.

IUCN Stresses Classification

1.2 Ecosystem degradation
2.1 Species mortality
2.3.2 Competition

IUCN Threats Classification

7.2 – Dams and water management/use
8.1 – Invasive non-native/alien species/diseases
11.3 – Temperature extremes
11.5 – Climate change & severe weather; other impacts

Ecosystem collapse

Alpine-subalpine Lakes collapse when they cease to be permanent water bodies. They are also defined as collapsed if their biota changes from characteristically cold-water adapted species to those that are found at lower altitudes, or if the timing of snow melt alters substantially.

1. If there was a significant reduction so there was not a visible water table, resulting in total loss of any permanency of water in the lakes. A change in hydrological conditions that leads to lakes moving from permanent to ephemeral or seasonal would also be considered a collapsed state. This may occur via a reduction in surface or groundwater, or a reduction in precipitation under climate change to the point where the lakes are unable to persist year-round. This change may occur with a minimum of one dry period within a year, representing a transition to a novel ecosystem type.
2. If the extent and duration of ice cover on the lake changes, resulting in changes to timing of water mixing and seasonal stratification.
3. If the water became very polluted via sedimentation and thus uninhabitable by the characteristic native biota.
4. If the composition of the aquatic biota changes to being dominated by non-native, invasive species, or dominated by native species characteristic of lower-elevation lakes due to the upward migration of littoral vegetation under a warming climate.

Ecosystem collapse in *Alpine-subalpine Lakes* is defined in this assessment as when*:*

1. The mapped distribution of lakes with permanent water declines to zero (100% loss) (Criteria A and B).
2. The average annual water level declines to 0 m (Criterion C)
3. Reduced length of snow cover season (season length is reduced by 50%; Criterion C)
4. Mean maximum surface water temperature exceeds 20 degrees Celsius (Criterion C)
5. Macroinvertebrate assemblage: The SIGNALT macroinvertebrate assemblage condition score is two thirds the value at which the assemblage is considered to be poor, equating to a SIGNALT score for 3.4 in Victoria, 3.1 in Tasmania and 1.9 in NSW and ACT (Criterion D). SIGNALT provides a condition score by combining sensitivity scores for each taxon found within the ecosystem to provide an overall assessment between 1 and 10, where 1 is a completely tolerant assemblage with no sensitive taxa and 10 is an assemblage that include taxa sensitive to environmental conditions.

Risk Assessment

Criterion A: Reduction in distribution

Summary

There may be future reductions in the distribution of *Alpine-subalpine Lakes* due to future climate change reducing the availability of water (Lough & Hobday 2011) (sub-criterion A2). Past changes in distribution have not been quantified but appear very unlikely to have exceeded a 30% reduction in the past 50 years (sub-criterion A1) or since European invasion (sub-criterion A3). However, no data are available that capture the actual distribution of *Alpine-subalpine Lakes*. Existing GIS layers, for example, define lakes by low points in topography rather than based on any assessment of the location of water in the landscape or other measure of the ecosystem type itself. Furthermore, these layers are not updated regularly and so do not capture any changes in distribution either. Hence, we assess Criteria A as **Data Deficient** for each of sub-criteria A1, A2 and A3.

Criterion B: Restricted distribution

Summary

The Extent of Occurrence (EOO) for *Alpine-subalpine Lakes* is **Least Concern** under Criterion B.

Methods

Under criterion B, the extent of occurrence (EOO; B1) and area of occupancy (AOO; B2) of alpine-subalpine lakes were determined using an existing national map product (National Surface Water Hydrology: Crossman & Li 2015; Geoscience Australia 2021). These data were checked against expert knowledge and available hydrological data and altered where necessary to best reflect on-ground information (i.e., whether lakes are natural or artificial). A complete description of the data and methods used to create the current distribution map for this ecosystem is provided in the main methods.

The EOO was calculated using a minimum convex polygon enclosing all mapped occurrences of alpine-subalpine lakes in Australia. The AOO was calculated based on the number of 10 x 10 km grid cells that contained the ecosystem. The number of threat-defined locations was based on fire and changes to water stratification under climate change the most important threats to alpine-subalpine lakes.

Assessment outcome

Sub-criterion B1

The Extent of Occurrence (EOO) of *Alpine-subalpine Lakes*, based on available mapping, is currently estimated at 69,552.35 km^2 (Figure 119). The risk status is **Least Concern** under sub-criterion B1. However, only a very small part of the distribution occurs on the mainland (0.28 km^2, or 0.28% of the current distribution). If the mainland extent (five lakes) were to locally collapse, the EOO would only encompass the Tasmanian distribution (which covered 97.7% of the distribution) and thus decline to an EOO of 30,486 km^2, equating to a status of Vulnerable.

Sub-criterion B2

Alpine-subalpine Lakes occupy 69 10x10 km grid cells (Area of Occupancy, AOO) (Figure 119), including 1 on the mainland and 68 on Tasmania. The risk status of the ecosystem is **Least Concern** under sub-criterion B2.

Sub-criterion B3

Alpine-subalpine Lakes may be threatened by the increase in large fires across the alpine regions of Australia. Fires can burn the surrounding vegetation, leading to runoff with ash, nutrients and fire-fighting foams polluting the water. *Alpine-subalpine Lakes* are narrowly distributed across the mainland (1 location) widely dispersed across Tasmania (> 5 locations). However, it seems unlikely that the pollution stemming from a single fire would cause collapse; several fires over a short period (~20 years) would likely be required to degrade the system enough to cause collapse. As a result, the risk status of the ecosystem type is **Least Concern** under sub-criterion B3.

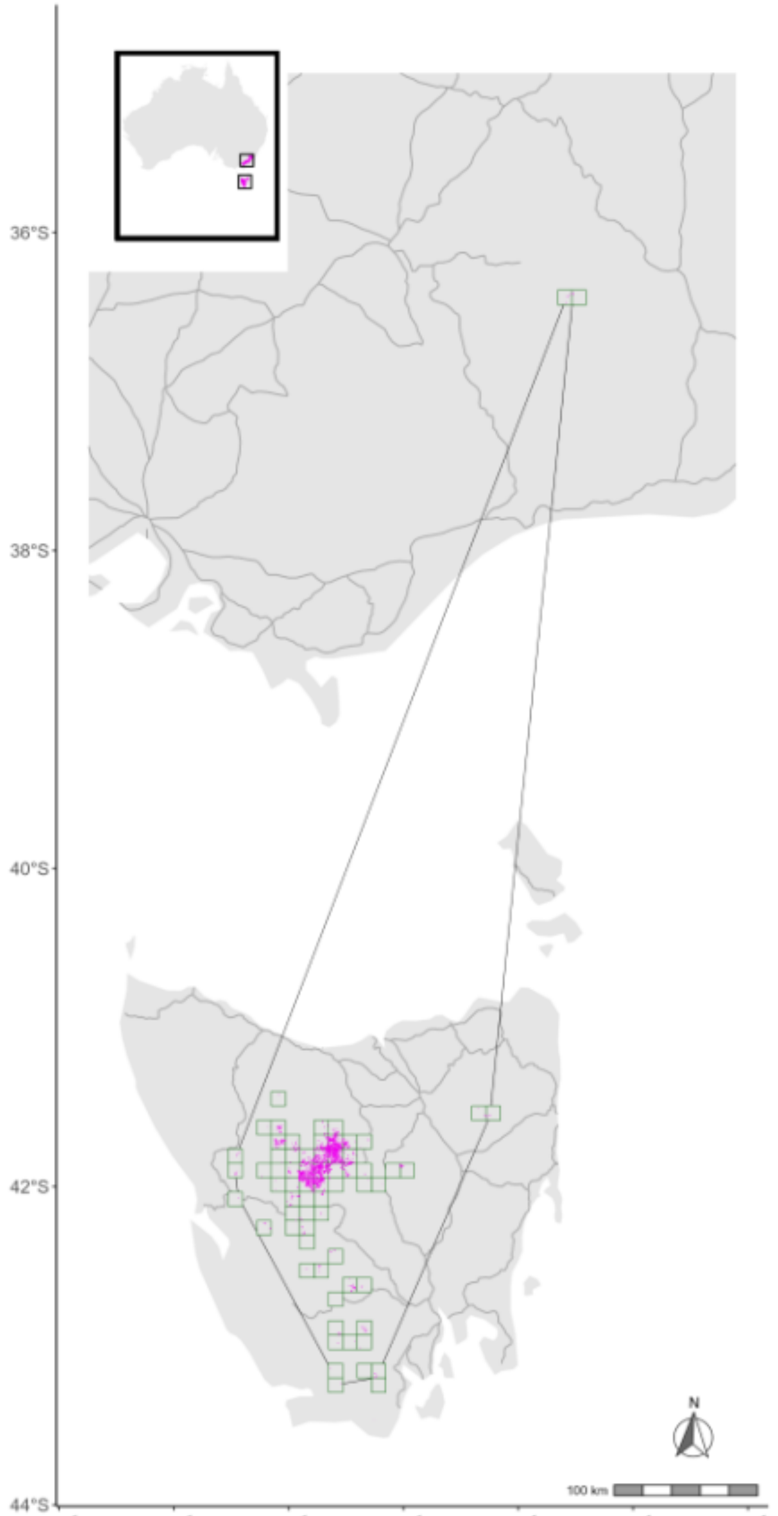


Figure 119. Map of *Alpine-subalpine Lakes* (magenta) showing EOO (black polygon) and AOO (light green squares).

Criterion C: Environmental degradation

Summary

Based on the available temporal and spatial hydrological data for *Alpine-subalpine Lakes*, there were no identified trends in annual average lake level in the recent past, while insufficient data exist for long-term (historical) analyses. Therefore, the status of the ecosystem is **Least Concern** under sub-criteria C1 and C2b for annual lake water level and **Data Deficient** under sub-criterion C3. The indicators length of the snow cover season and water temperature were Data Deficient for all sub-criteria.

Identification of abiotic indicator

We examined the relevance and data availability/quality for three abiotic indicators to assess the risk of collapse from environmental degradation:

- *Annual lake water level*: a measure of the volume of water contained within the lake in each year
- *Water temperature*: as a measure of the temperature (degrees Celsius) of the waterflow
- *Length of the snow cover season*: a measure of how long snow persists in the alpine and subalpine region

Indicator: Annual water level

Relevance to ecosystem function

Alpine-subalpine Lakes are defined as being permanent lake systems. This is associated with the presence of a water table above the surface. Changes to the water level may lead to the system transitioning from permanent to intermittent or ephemeral lakes, or to a lack of any permanent standing water (Boulton et al. 2014). An accepted definition of a 'permanent or near permanent' waterbody is one that experiences predictable flooding, where the annual input of water is greater than the losses in 9 out of 10 years, or one that is predictably filled, with annual inflows exceeding minimum annual loss 90% of the time (Boulton et al. 2014). While near permanent lakes can dry during extreme drought, any increase in the regularity of that drying, or shift to seasonal or frequent drying fundamentally changes the character of the lake, potentially reducing the suitability of *Alpine-subalpine Lakes* for many characteristic species as these systems often support diverse aquatic life, much of which cannot tolerate desiccation (Boulton et al. 2014). As a result, changes in the annual average water level are directly related to the continued status of *Alpine-subalpine Lakes* as near permanently inundated systems. If annual average water levels decline through time, that suggests that the lake is drying and is more likely to be intermittent or ephemeral.

Data availability and quality

Recent change (C1)

We used the R package ‘hydrogauge’ (Holt 2025) to locate all available water level stations listed by the Bureau of Meteorology (BOM 2025). These locations were then intersected with the Shuttle Radar Topography Mission (SRTM) elevation data (CGIAR-CSI 2018) using the ‘geodata’ R package to identify stations situated above the elevation threshold for alpine or subalpine zones within each relevant state. In cases where fewer than five stations were found in these high-elevation areas – such as in NSW – we relaxed the elevation threshold to ensure a minimum of five stations per state (it is noted where we have used this expanded set for clarity). A minimum of five was chosen as it represents a commonly accepted sample size for detecting relationships. This process resulted in 9 stations located above the elevation threshold, with 8 in Tasmania and a single station in New South Wales (NSW) (Figure 120). None were identified about the elevation threshold in ACT. An additional 9 stations from the ACT and NSW, which fell below the threshold, were included in a secondary analysis to explore the effect of having limited representation from truly alpine sites. It is also important to note that, while the definition of Alpine-subalpine Lake ecosystems is confined to natural lake systems, many of the gauging stations for water levels are on managed waterways. Of the stations identified, only two appeared to be truly natural – Lakes Augusta and Fenton in Tasmania. The others appear to be managed. Nonetheless, we have included them to illustrate trends in water levels because managed lakes and reservoirs will also show declines in water level through time if groundwater and snow levels are declining. Thus, we considered these stations to be representative of lake water levels in alpine and subalpine environments across the three states.

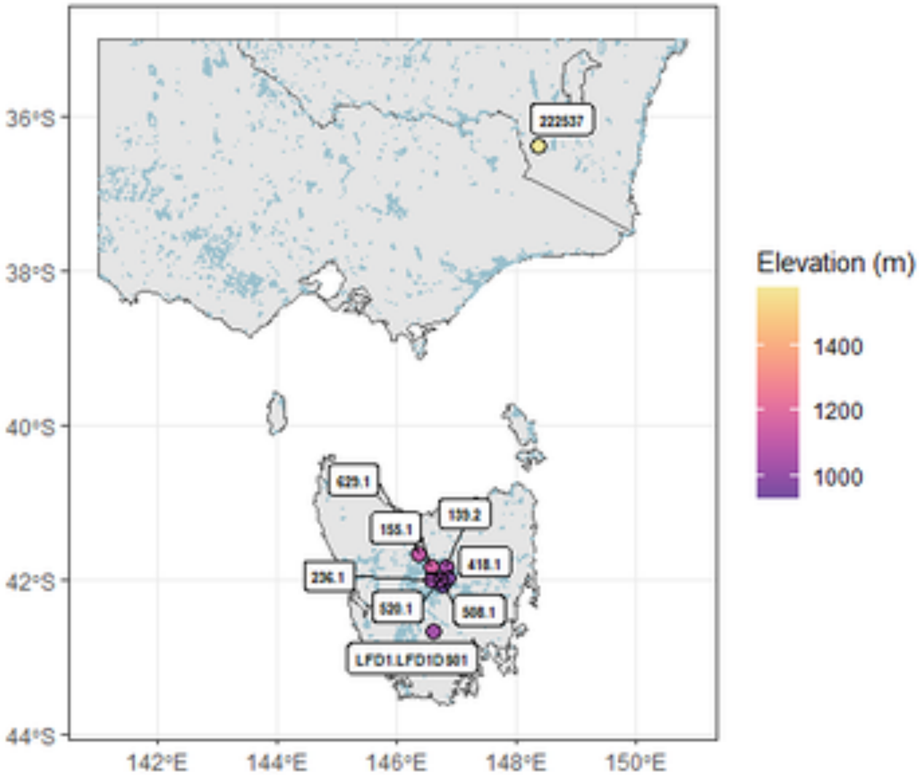


Figure 120. Station locations used to assess trends in average annual water level over the historical record including 9 gauges (see labels for gauge IDs) in the alpine/subalpine zone across two states.

Seven of the 9 alpine/subalpine stations had records that extended for at least 50 years. Three of the additional 9 stations at lower elevations also had records of at least 50 years. For ease of analysis, all years of record were included for all stations.

Future change (C2)

We extrapolated the trend in water levels to estimate the trajectory of decline for *Alpine-subalpine Lakes* across the two sets of stations (9 alpine/subalpine and 18 to include at least five in each state).

Historical change (C3)

Data were unavailable to assess historical change in hydrologic indicators.

Selection of collapse threshold

The drying of Alpine-subalpine Lakes will have a large impact on the character of those lakes as many freshwater biota are restricted to alpine waterways because they cannot tolerate warming or drying (Boulton et al. 2014). Given this intolerance to drying, we determined that a lake level of 0 m would constitute collapse. Thus, we define collapse as when the annual average water level of a lake reached 0 m.

Calculation of initial and present/future values

Recent change (C1) and Future change (C2)

Average and minimum annual lake levels were calculated for each year in the historical time series for each station using the R statistical environment. Because of the very large variance in the long-term average annual lake level (Figure 121), we standardised all average annual lake levels by the long-term average level for that lake (Figure 122). We also removed any stations that had fewer than 20 years of record, as long-term averages would not be meaningful. For the alpine-subalpine stations, this resulted in a sample size of 8 stations, which was too few to be meaningful. Therefore here, we elected to analyse the larger set including those below the alpine-subalpine thresholds in each state (2 stations removed due to < 20 years of record, leaving 16 stations). We fit a generalised linear model to the data, assessing average annual water level as a function of year and elevation, with station included as a random variable.

There was no statistically significant trend in change in lake level relative to the long-term average across the suite of stations through time (**Error! Reference source not found.**122) or with elevation, and so no extrapolation was undertaken to assess future change.

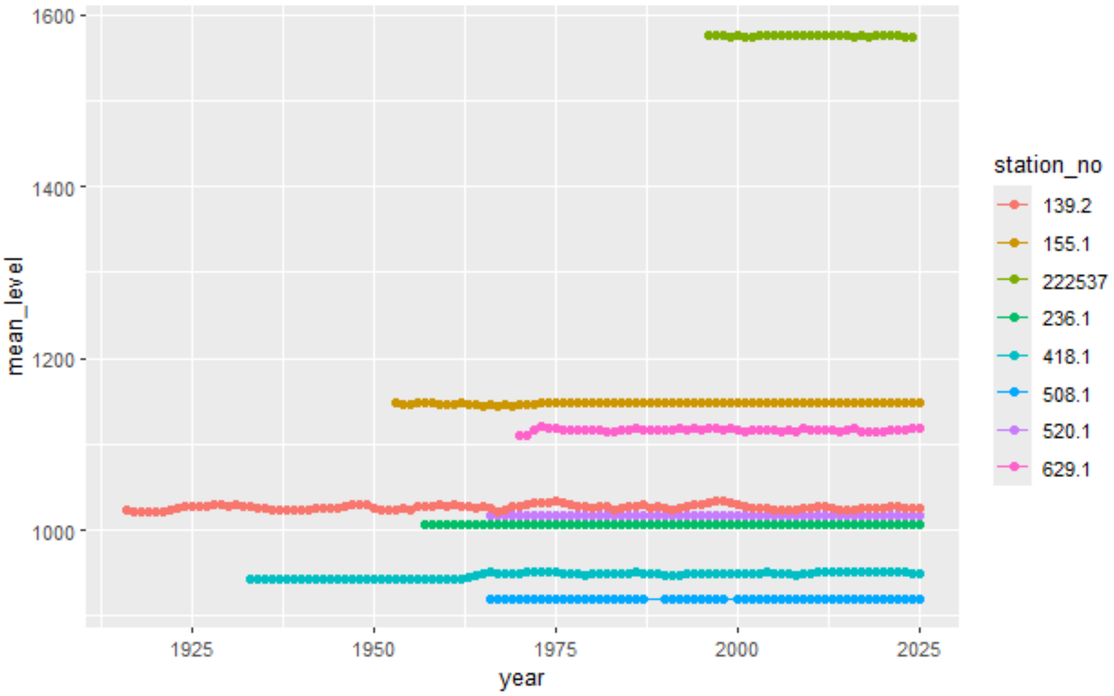


Figure 121. Trend in average annual lake level over the historical record for 8 stations in the alpine/subalpine zone across two states which had more than 20 years of record (noting that the period of data for each station varies).

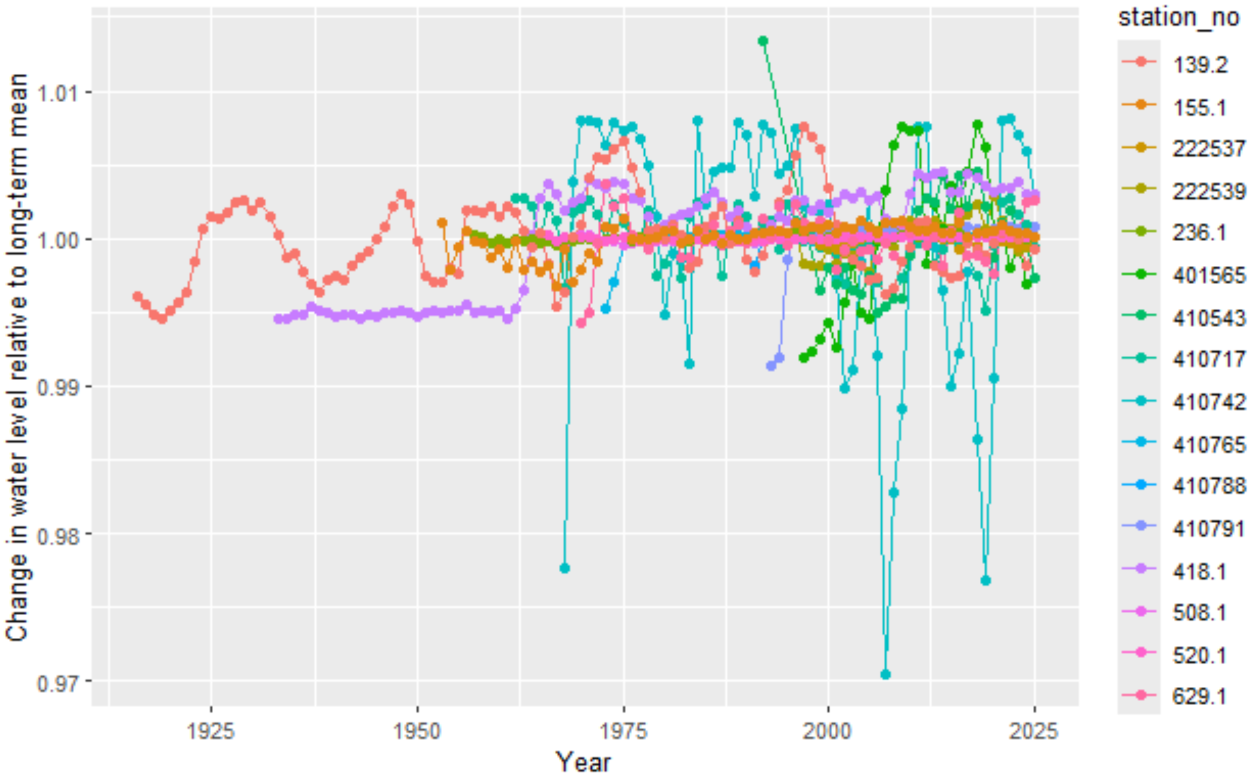


Figure 122. Trend in change in annual lake level relative to the historical mean for 16 stations that had more than 20 years of record, expanded below the alpine/subalpine zone across three states (noting that the period of data for each station varies). Analyses were conducted on the available length of record for each station, where that station had more than 20 years of record (range 1914-2025).

Historical change (C3)

Data were unavailable to assess this sub-criterion.

Calculation of severity and extent

Recent change (C1) and Future change (C2b)

There was no trend in change in lake level relative to the long-term average across the historical record. As a result, the risk status is **Least Concern** under sub-criterion C1 and sub-criterion C2b.

Historical change (C3)

There were insufficient data to assess historical changes in baseflow. The risk status is **Data Deficient** under sub-criterion C3.

Indicator: Water temperature

Water temperature is a key factor for the aquatic biota present in *Alpine-subalpine Lakes* as it impacts the timing of invertebrate breeding and provides an ecological barrier that allows endemic species to outcompete invasives (i.e., because invasive species are often unable to tolerate the cold temperatures). Lake water temperature is largely driven by groundwater temperatures, although is also influenced by light penetration and air temperature (Closs et al. 2009; Boulton et al. 2014). Measuring change in water temperature would require monitoring groundwater across the ecosystem's distribution, however, these data are not available. The ecosystem type is therefore assessed as **Data Deficient** for this indicator.

Indicator: Length of snow cover season

Relevance to ecosystem function

Alpine-subalpine Lakes are groundwater dependent and fed by snowmelt (Boulton et al. 2014). Spring snowmelt is the primary source of water for this ecosystem type (Čiamporová-Zat'ovičová 2011). Declines in snow cover and earlier melting of snow may reduce the water level in this ecosystem.

Selection of collapse threshold

We considered that the ecosystem type would collapse is the length of the snow cover season declined by 50%.

Data availability and quality

There were insufficient data to assess this indicator, and this it is therefore Data Deficient.

Criterion D: Disruption of biotic processes and/or interactions

Summary

Based on the available macroinvertebrate assemblage condition data for *Alpine-subalpine Lakes*, there was a statistically significant decline in condition over the historical record in NSW and Tasmania (noting a similar decline in lakes below the threshold in ACT). Over a 50-year period, this decline was predicted to reach the collapse threshold. Therefore, the status of the ecosystem is **Endangered** under sub-criteria D1, **Critically Endangered** under sub-criterion D2b and **Data Deficient** under sub-criterion D3.

Identification of biotic indicator

We examined the relevance and data availability/quality for one biotic indicator, selected based on our conceptual model, to assess the risk of collapse from disruption of biotic processes and/or interactions:

- SIGNALT macroinvertebrate assemblage condition score

Indicator: SIGNALT score

Relevance to ecosystem function

While the macroinvertebrate fauna of *Alpine-subalpine Lakes* tends to be poorly described, there are numerous species, including stoneflies, mayflies and others that are likely to be restricted to high mountain streams (Williams et al. 2014). Additionally, aquatic habitats in Tasmania and other alpine regions are likely affected by summit traps, where isolated populations and limited dispersal capabilities hinder movement to alternative suitable habitats (Boulton et al. 2014). This is especially true given that many alpine species have evolved with reduced or absent wings as a common adaptation (Campbell et al. 1986). Further, invertebrates in alpine lakes have been identified as effective indicators of temperature changes elsewhere (Čiamporová-Zat'ovičová 2011). As a result, macroinvertebrate communities serve as a dependable indicator of biotic functioning.

Due to the limited availability of detailed information on distribution and ecology, evaluating the overall condition of macroinvertebrate communities offers a practical approach to assessing biological ecosystem function. SIGNAL 2 is a well-established tool for gauging the condition of Australian aquatic macroinvertebrate assemblages (Chessman 2003). The method assigns scores to genera or families of invertebrates based on their known tolerance to pollution and dissolved oxygen levels (Chessman 2003). A modified version, known as the SIGNALT index, has been developed for use by citizen scientists.

Data availability and quality

Recent change (D1)

SIGNALT scores derived from macroinvertebrate assemblage data were obtained from the Waterbug Data Portal on the National Waterbug Blitz website (National Waterbug Blitz Team 2025). The data were collected through combined efforts of the Environment Protection Agency and community groups in New South Wales, the Australian Capital Territory, and Tasmania. We identified sampling sites located within a 5-kilometre buffer of the stations chosen under Criterion C to ensure the relevance of selected data. As for Criterion C, we used two sets – those that fell above the elevation threshold for the alpine/subalpine region in each state (14 sampling locations) and a second larger set associate with the inclusion of additional stations at lower elevations to ensure that at least 5 stations were available per state (23 sampling locations). We assumed that these sampling locations were representative of the condition of macroinvertebrate assemblages in *Alpine-subalpine Lakes* across the three states, although ACT only had one sampling locations that met the selection criteria.

Within the alpine-subalpine stations, records were extremely patchy, with most of the 14 sampling sites having data for one to two years. One sampling location had been sampled six times. This was the maximum available data for any of the alpine-subalpine sampling locations. Data were recorded between 1995 and 2012, with additional sampling in 2022 and 2023 for numerous locations. A similar pattern emerged with the additional sampling locations. Again, most locations had data for a small number of years. Six years was the maximum number of samples for any location. Here, data records occurred between 1994 and 2023.

Data quality may vary due to the combination of samples collected by EPA staff and citizen scientists. Nevertheless, the Waterbug Blitz is a well-established citizen science program that provides consistent, high-quality training and implements rigorous quality assurance procedures for all data collected. Furthermore, most of the available data pre-date the Waterbug Blitz and so were collected by EPA staff. Despite not being specifically designed for alpine and subalpine regions, we are confident that the overall data quality is adequate to support the findings presented in this report.

Future change (D2)

To estimate the trajectory of decline for *Alpine–Subalpine Lakes*, we extrapolated trends in SIGNALT scores describing the relative condition of macroinvertebrate assemblages at sampling sites located within 5 kilometres of the stations identified under Criterion C. As with Criterion C, we used two data sets to maximise coverage: one consisting of 14 sampling locations within the alpine/subalpine region, and a broader set of 23 locations within 5 km of a larger group of stations, selected to ensure a minimum of five sites per state.

Historical change (D3)

Data were unavailable to assess historical change in the condition of macroinvertebrate assemblages.

Selection of collapse threshold

Existing evaluations of the relative health of macroinvertebrate assemblages use thresholds in SIGNALT score to define poor condition (National Waterbug Blitz Team 2025), and these differ by state: 4.7 in Tasmania and 2.9 in New South Wales. To establish a collapse threshold, we chose a value equal to two thirds of the poor condition threshold, reflecting a significant decline beyond the point where the assemblage is already considered poor. Accordingly, the collapse thresholds for the SIGNALT scores are 3.1 for Tasmania and 1.9 for New South Wales.

Calculation of initial and present/future values

Recent change (D1) and Future change (D2)

The SIGNALT score for each alpine-subalpine sampling site in the time series was analysed using the R software environment. We applied a generalized linear model to examine SIGNALT scores as a function of year, elevation, and state, including the interaction between elevation and state. This analysis was conducted for both the 14 sampling locations within the alpine/subalpine region and the broader set of 23 sites (Figure 123). There is no evidence that a decline in SIGNALT score would necessarily follow a linear model, but in the absence of evidence of a different relationship, we have used a linear relationship as an assumption.

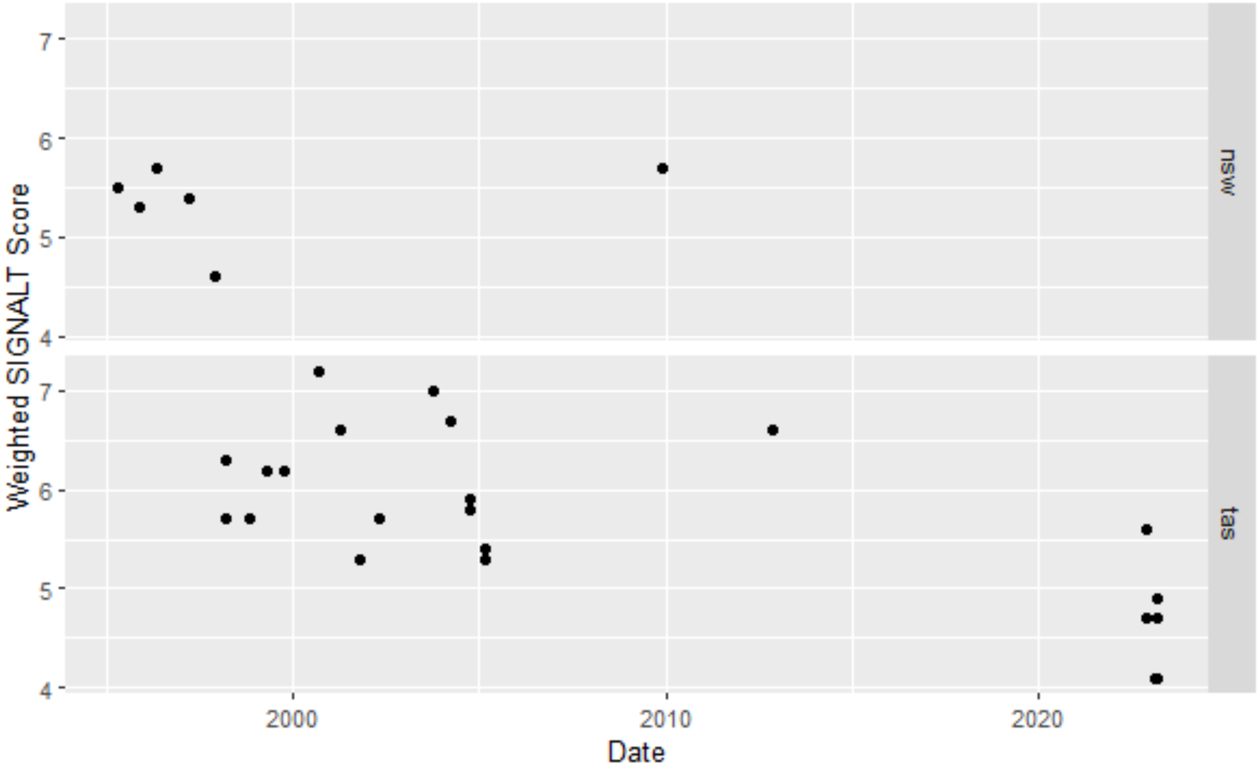


Figure 123. Observed SIGNALT scores over the historical record for 14 sampling locations in the alpine/subalpine zone across three states. Analyses were conducted on the available length of record for each sampling location (range 1994-2023).

Historical change (D3)

Data were unavailable to assess this sub-criterion.

Calculation of severity and extent

Recent change (D1) and Future change (D2b)

There was a highly statistically significant decline in SIGNALT score through time ($F_{1,34} = 15.82$, $p < 0.001$). SIGNALT score also declined significantly with increasing elevation ($F_{1,1} = 15.18$, $p < 0.001$; Figure 124).

When analysing the additional sites below the relevant elevation thresholds in each state, the overall finding of a statistically significant decline in SIGNAL score through time persisted. There was also a significant effect of state but elevation was not a significant factor in the analysis, suggesting the effect of elevation may be confined to truly alpine-subalpine locations.

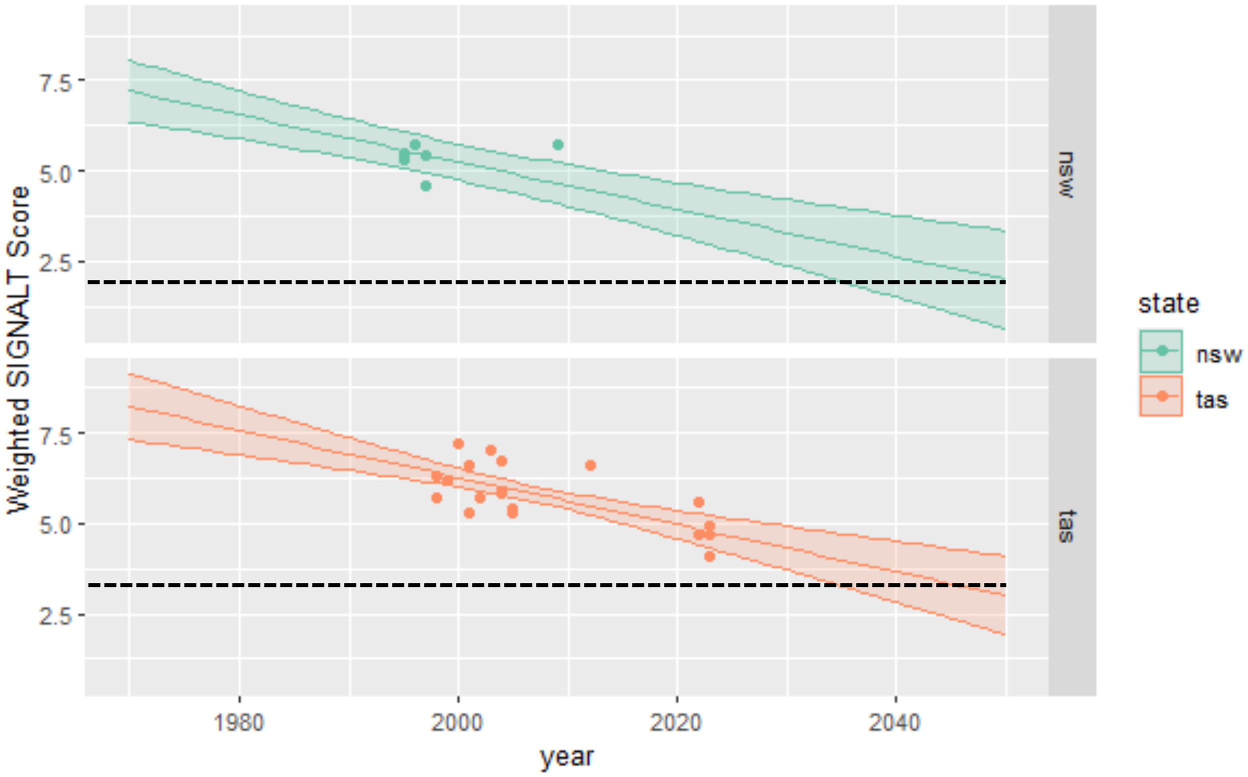


Figure 124. Trend in SIGNALT score extrapolated for the period of 1975 to 2050, including the observed SIGNALT scores over the historical record for 14 sampling locations in the alpine/subalpine zone across three states. Analyses were conducted on the available length of record for each sampling location. Relevant collapse thresholds for each state are showing with a dotted black line.

To calculate severity of change, we first calculated the average change per year in SIGNALT score over a 50-year period, based on the calculated trend. For the period 1970-2020 (relative to sub-criterion D1), the relative severity is 100 x (8.51 – 3.96)/(8.51 – 3.1) = 84.1% in NSW and 100 x (9.53 – 4.98)/(9.53 – 1.9) = 59.6% in Tasmania. This equates to a risk status of **Endangered** under sub-criterion D1.

For the period 2000-2050 (relative to sub-criterion D2b), the relative severity of change is 100 x (5.25 – 1.99)/(5.25 – 1.9) = 97.3% in NSW and 100 x (6.27 – 3.01)/(6.27 – 3.1) = 102.9% in Tasmania. This equates to a risk status of **Critically Endangered** under sub-criterion D2b. The findings were similar for both sub-criteria when additional sampling sites below the relevant elevation thresholds in each state were included.

Historical change (D3)

There were insufficient data to assess historical changes in SIGNALT score. The risk status is **Data Deficient** under sub-criterion D3.

Criterion E: Quantitative analysis of probability of collapse

Criterion E requires a stochastic simulation model capturing the key features and processes of alpine-subalpine lakes to estimate the probability of collapse with the next 50 to 100 years based on likely future threat scenarios. The general lake model is a hydrodynamic model designed to provide a strandardised approach for simulations of lakes, reservoirs and water bodies (Christianson et al. 2019). Though it would need to be heavily adapted to suit alpine lakes to account for the unique characteristics that differentiate them from non-alpine lakes, it could be used to assess potential risks from multiple threats and the likelihood of those risks leading to a collapsed state, noting that different types of lakes (e.g., monomictic versus dimictic) would need to be modelled separately. The data required to update the model are not yet available, however should this change, it would be a potential indicator in future assessments. Due to a lack of data available to adapt the general lake model, the risk status is **Data Deficient** under Criterion E.

Assessment outcomes

Seven (47%) ecosystem types were classified as threatened, including one as Endangered and five as Vulnerable. This equates to 553.47 km^2 (7.25% of the study area) covered by threatened ecosystem types. Three ecosystem types were assessed as Near Threatened, three as Least Concern, and one ecosystem type was assessed as Data Deficient.

In the Terrestrial Biome, three ecosystem types were assessed as Least Concern, three as Near Threatened, four as Vulnerable and one as Endangered (Table 41). In the Terrestrial-Freshwater Biome, two ecosystem types were assessed as Least Concern and the other Data Deficient. In the Freshwater Biome, one ecosystem type was assessed as Engendered and other Critically Endangered.

Most ecosystem types were listed as threatened or near threatened due to declines in distribution or having a restricted distribution and ongoing threats, including future declines in environmental suitability under climate change, altered fire regimes, and hard-hooved animals. However, two ecosystem types were listed as threatened due to declines in integrity: *Snowpatch Herbfield* due to snowmelt date and *Alpine-subalpine Streams* due to changes in the macroinvertebrate assemblage.

Analysis showed that for several ecosystem types, fire regimes are not predicted to increase to a frequency likely to cause large declines in the next 50 years. Further, fire frequency has increased in recent years in *Subalpine Woodland and Forest* and *Alpine-subalpine Closed Heath,* contributing towards listing as Near Threatened.

Sufficient data were not available to assess collapse risk due to changes in the biotic features and processes for many ecosystem types, across Criteria C and D, a key limiting factor in producing reliable assessments of future risk. There were insufficient data to assess risk in most Criteria for the freshwater ecosystem types. Further we were only able to assess Criteria E for *Alpine-subalpine Streams*. There was a potential suitable simulation model for *Alpine-subalpine Lakes,* but there were insufficient data to parameterise this model suitably to assess Criterion E.

Dominant threats (most commonly list as a threat among ecosystem types) were climate change-related threatening processes, including temperature extremes, drought, and changes to fire regimes. trampling, herbivory and wallowing by invasive ungulate species, and human land uses, including infrastructure, recreational activities, and farming were also common threats among many ecosystem types. We summarise assessment outcomes for each ecosystem type in Table 41.

Table 41. Relationship between the ecosystem units in this project, the IUCN global ecosystem typology (Keith et al. 2022a), typology of alpine and subalpine ecosystems of Australia (Venn et al. 2017), and the IUCN RLE Assessment outcomes.

Ecosystem Units	**IUCN Global Ecosystem Typology**	**IUCN RLE Assessment** (values in parentheses show plausible bounds)
Tasmanian Palaeoendemic Forest and Woodland	T2.3 Oceanic temperate rainforests	**Vulnerable** (Near Threatened – Endangered)
Alpine-subalpine Woodland and Forest	T4.4 Temperate woodlands	**Near Threatened**
Alpine-subalpine Open Grassy Heathland and Herbfield	T6.4 Temperate alpine meadows and shrublands	**Least Concern** (Least Concern – Near Threatened)
Alpine-subalpine Closed Heath	T6.4 Temperate alpine meadows and shrublands	**Least Concern**
Coniferous Heath	T6.4 Temperate alpine meadows and shrublands	**Vulnerable** (Vulnerable – Endangered)
Fjaeldmark/feldmark	T6.4 Temperate alpine meadows and shrublands	**Vulnerable**
Snowpatch Herbfield	T6.4 Temperate alpine meadows and shrublands	**Endangered** (Vulnerable – Endangered)
Cushion Moorland	T6.4 Temperate alpine meadows and shrublands	**Vulnerable**
Tasmanian Alpine Heath	T6.4 Temperate alpine meadows and shrublands	**Near Threatened** (Least Concern – Near Threatened)
Tasmanian Alpine Sedgeland	T6.4 Temperate alpine meadows and shrublands	**Near Threatened** (Least Concern – Near Threatened)
Alpine-subalpine Damp Valley Grassland and Rushland	T6.4 Temperate alpine meadows and shrublands	**Least Concern**
Alpine *Sphagnum* Bog and Associated Fen (nationally)	TF1.5 Boreal, temperate and montane peat bogs	**Least Concern** (Least Concern – Near Threatened)
Alpine *Sphagnum* Bog and Associated Fen (mainland)	TF1.5 Boreal, temperate and montane peat bogs	**Vulnerable** (Vulnerable – Endangered)
Alpine *Sphagnum* Bog and Associated Fen (Tasmania)	TF1.5 Boreal, temperate and montane peat bogs	**Endangered** (Endangered – Endangered)
Alpine-subalpine Fen	TF1.5 Boreal, temperate and montane peat bogs	**Data Deficient**
Alpine-subalpine Streams	F1.3 Freeze-thaw rivers and streams	**Endangered**
Alpine-subalpine Lakes	F2.4 Freeze-thaw freshwater lakes	**Critically Endangered**

Recommendations

Below we outline recommendations based on the development of a typology of Australian Alpine ecosystem types, the mapping of these ecosystems, and the Red List of Ecosystem assessment outcomes.

Develop monitoring of abiotic and biotic indicators

Many of the ecosystem types had insufficient data to assess progress towards collapse for many abiotic and biotic indicators. These indicators, discussed in Criterion C and D, are essential to understand the ecological integrity of ecosystems, to quantify any degradation, and to assess and achieve Goal A and Target 2 of the Global Biodiversity Framework. Outcomes from this report suggest that the development of monitoring to support assessment would be useful for a range of indicators. In Table 42 we list the indicators that would benefit from the establishment of data collection processes.

Table 42. Indicators that require additional collection, monitoring and assessment of data to understand ecosystem integrity.

Ecosystem	**Criterion**	**Indicator**
Subalpine Palaeoendemic Forest and Woodland	C	Fire occurrence
	D	Tree and shrub cover
Alpine-subalpine Woodland and Forest	C	Soil moisture
	D	Dead tree density or abundance
	D	Weed cover
	D	Exotic herbivore density
Alpine-subalpine Closed Heath	D	Abundance of exotic ungulates
	D	Grass and forb cover
Coniferous Heath	D	Conifer vegetation cover
Fjaeldmark	C	Growing degree days
	C	Mean daily minimum temperature of the coldest month
	D	Vegetation Cover
Snowpatch herbfield	C	Snowpack depth
	D	Plant phenology
Cushion Moorland	C	Indicator pertaining to temperature and desiccation
	C	Indicator related to fire frequency
	D	Cushion plant cove
	D	Bare ground cover:
Tasmanian Alpine Heath	D	Plant species composition:
	D	Plant phenology
Tasmanian Alpine Sedgeland	C	Post-fire soil erosion
	C	The number of growing degree day
	D	Monocot cover
	D	Shrub cover
Alpine-subalpine Damp Valley Grassland and Rushland	C	Bare Ground cover
	C	Soil moisture
	C	Number of dry months
	D	Plant species composition

Alpine Sphagnum Bog and Associated Fen	C	Number of dry months
	C	Fire frequency
	D	Vegetation Cover
Alpine-subalpine Lakes	C	Length of snow cover season
	C	Surface water temperature

Undertake targeted data collection to understand critical thresholds and indicators

There were a number of threatening processes to ecosystem types for which we were unable to determine thresholds of indicators of degradation, due to limited knowledge on what level of change constitutes a threat to this ecosystem. For example, Cushion Moorland is susceptible to environmental degradation due to climate change, due to increased frequency of high temperatures, droughts and fires, However, the temperature and desiccation conditions associated with cushion dieback have not been quantified, nor has the response of cushions to recurring fires. This meant we were unable to assess progress towards indicators of collapse. Below we list the following information that requires targeted study to understand ecosystem responses and determination of indicators and their thresholds:

- Cushion Moorland: Temperature and desiccation responses
- Cushion Moorland: Fire response
- Tasmanian Alpine Sedgeland: Level of erosion response
- Tasmanian Alpine Sedgeland: Temperature responses
- Alpine Sphagnum Bog and Associated Fen: Fire response

Develop available methodologies appropriate to assess Criterion E

We were only able to assess Criterion E for *Alpine-subalpine Streams*. The inability to assess Criterion E for most ecosystem types was primarily due to a lack of methods suitable for assessing the probability of collapse, such as a stochastic model. To be able to assess the ecosystem risk in future, the development of stochastic modelling methods to assess the probability of collapse will be important.

Revise ecosystem classification approaches and related spatial products

The reliability of the assessments was limited by low quality and inconsistent state-level ecosystem maps. The development of a national typology was also constrained by the requirement to use existing classifications under the Environment Protection and Biodiversity Conservation Act 1999 (e.g., *Alpine Sphagnum Bog and Associated Fen*). Revising the classification approach is advised in future reassessments of these ecosystem types, especially the water-dependent system.

Develop spatial tools and models to assess current and future fire in Tasmanian landscapes

We were only able to assess future fire related outcomes in the mainland of Australia. However, much of Australian Alpine ecosystems are in Tasmania. The development and availability of future fire simulations and data products for Tasmania would greatly improve the ability to assess fire related threats to Australian Alpine ecosystems.

Take action to protect Vulnerable, Endangered and Critically Endangered Ecosystems

Alpine-subalpine Lakes was assessed as Critically Endangered, *Snowpatch Herbfield, Alpine Sphagnum Bog and Associated Fen* (mainland and Tasmanian extents separately) and *Alpine-subalpine Streams* were assessed as Endangered, *Tasmanian Palaeoendemic Forest and Woodland*, *Coniferous Heath, Fjaeldmark/feldmark, Cushion Moorland* and *Alpine Sphagnum Bog and Associated Fen* (nationally) were assessed as Vulnerable. Targeted conservation actions should be developed and implemented to protect these ecosystems, as per Australia's obligations to protect and restore degraded ecosystems under the Global Biodiversity Framework.

Management actions to mitigate the spread of invasive ungulates and their impacts

Many of the ecosystem types within Australian alpine and subalpine ecosystems are threatened by invasive ungulates. Work to manage the spread of invasive ungulates would minimise the damage caused to Australian Alpine ecosystems through trampling, browsing and wallowing. Such management efforts could include ungulate removal actions such as aerial culling (Driscoll et al. 2019).

Monitor climate related threats such as temperatures, droughts and increased fire frequencies and take suitable actions

Australian alpine and subalpine ecosystems are threatened by climate change driven temperature, precipitation and fire frequency changes. While many of climate change driven threats are out of localised management control, monitoring prolonged episodes of drought, high temperatures, or approaching fires and taking responsive action actions will help to minimise these threats. For example, using fire suppression in recently burnt areas where fire is fast approaching.

References

AALC. 2006. The Australian Alps rehabilitation manual: A guide to ecological rehabilitation in the Australian Alps. Available from https://theaustralianalps.files.wordpress.com/2013/12/rehabilitation06.pdf.

ABARES. 2018. Forests of Australia (2018). Canberra. Available from https://www.agriculture.gov.au/abares/forestsaustralia/forest-data-maps-and-tools/spatial-data/forest-cover.

Abram N, Henley B, Gupta A, Lippmann T, Clarke H, Dowdy A, Sharples J, Nolan R, Zhang T, Wooster M. 2021. Connections of climate change and variability to large and extreme forest fires in southeast Australia. Communications Earth & Environment **2**:1–17.

Abrams J, Greiner M, Schultz C, Evans A, Huber-Stearns H. 2021. Can Forest Managers Plan for Resilient Landscapes? Lessons from the United States National Forest Plan Revision Process. Environmental Management **67**.

ACT Government. 2018. ACT Vegetation Map 2018. Available from https://data.actmapi.act.gov.au/arcgis/rest/services/data_extract/Environment/MapServer/9 (accessed February 4, 2021).

Adrian R, Walz N, Hintze T, Hoeg S, Rusche R. 1999. Effects of ice duration on plankton succession during spring in a shallow polymictic lake. Freshwater Biology **41**:621–634.

Ahyong ST. 2016. The Tasmanian mountain shrimps, Anaspides Thomson, 1894 (Crustacea, Syncarida, Anaspididae). Records of the Australian Museum **68**:313–364.

Alexander JM et al. 2016. Plant invasions into mountains and alpine ecosystems: current status and future challenges. Alpine Botany **126**:89–103.

Annandale B, Kirkpatrick JB. 2017. Diurnal to Decadal Changes in the Balance between Vegetation and Bare Ground in Tasmanian Fjaeldmark. Arctic, Antarctic, and Alpine Research **49**:473–486.

Anon. 2003. The impacts of bushfires following a flash flood event in the catchment of the Ovens River: Ovens Catchment Report Ovens Catchment Report. EPA Victoria, Melbourne.

ARCC. 2020. Victorian alpine resorts end of season report: Winter 2019. Available from https://www.arcc.vic.gov.au/uploads/publications-and-research/Winter Report 2019_FINAL.pdf.

ARMB. 2011. Recovery Plan for the Mountain Pygmy-possum Burramys parvus on Mount Buller, Victoria 2011 to 2016, upublished report to the Mount Buller and Mount Stirling Alpine Resort Management Board, Mt Buller Victoria.

Armstrong R, Turner K, McDougal K, Rehwinkel R, Crooks J. 2013. Plant communities of the upper Murrumbidgee catchment in New South Wales and the Australian Capital Territory. Cunninghamia **13**:125–265.

Ashton HD, Williams RJ. 1989. Dynamics of the sub-alpine vegetation in the Victorian region. Pages 143–68 in R. Good, editor. The Scientific Significance of the Australian Alps: The Proceedings of the First Fenner Conference on the Environment. Australian Alps National Parks Liaison Committee.

Atkin OK, Collier DE. 1992. Relationship between soil nitrogen and floristic variation in late snow areas of the Kosciuszko alpine region. Australian Journal of Botany **40**:139–49.

Auld J, Everingham SE, Hemmings FA, Moles AT. 2022. Alpine plants are on the move: Quantifying distribution shifts of Australian alpine plants through time. Diversity and Distributions:1–13.

Australian Government Department of Agriculture Water and Environment. 2020. National Vegetation Information System. Canberra, ACT.

Baines G, Webster M, Cook E, Seddon J. 2013. The vegetation of the Kowen, Majura and Jerrabomberra districts of the ACT, Technical Report 28.

Ballantyne M, Pickering CM. 2015a. Recreational Trails as a Source of Negative Impacts on the Persistence of Keystone Species and Facilitation. Journal of Environmental Management **159**:48–57. Available from https://doi.org/10.1016/j.jenvman.2015.05.026.

Ballantyne M, Pickering CM. 2015b. Shrub facilitation is an important driver of alpine plant community diversity and functional composition. Biodiversity and Conservation **24**:1859–1875.

Ballantyne M, Pickering CM, McDougall KL, Wright GT. 2014a. Sustained impacts of a hiking trail on changing Windswept Fjaeldmark vegetation in the Australian Alps. Australian Journal of Botany **62**:263–275.

Ballantyne M, Pickering CM, McDougall KL, Wright GT. 2014b. Sustained Impacts of a Hiking Trail on

Changing Windswept Feldmark Vegetation in the Australian Alps. Australian Journal of Botany **62**:263. Available from https://doi.org/10.1071/BT14114.

Banks JC. 1989. A history of forest fire in the Australian Alps. Pages 265–80 in R. Good, editor. The Scientific Significance of the Australian Alps. Australian Alps National Parks Liaison Committee, Canberra.

Barrett S, Yates CJ. 2015. Risks to a mountain summit ecosystem with endemic biota in southwestern Australia. Austral Ecology **40**:423–432.

Barrow M. D., Costin A. B., Lake P. 1968. Cyclical Changes in an Australian Fjaeldmark Community. Journal of Ecology **56**:89–96.

Bartley R, Rutherfurd I. 2005. Measuring the reach-scale geomorphic diversity of streams: Application to a stream disturbed by a sediment slug. River Research and Applications **21**:39–59.

Bayly IAE. 1970. A note on the zooplankton of the Kosciuszko region. Australian Society of Limnology Bulletin **3**:25–28.

Bayly IAE, Williams WD. 1973. Inland Waters and their Ecology. Longman Australia Pty Ltd, Camberwell, Victoria.

Bear R, Hill W, Pickering CM. 2006. Distribution and diversity of exotic plant species in montane to alpine areas of Kosciuszko National Park. Cunninghamia **9**:559–70.

Bennett SB, Mansergh I, Berwick S, Sandiford K, Wellington C. 1991. The Potential Effect of the Enhanced Greenhouse Climate Change on Selected Victorian Fauna. Heidelberg.

Benzie JA. 1984. Zooplankton of an Australian high alpine lake, Lake Cootapatamba, Kosciusko range. Marine and Freshwater Research **35**:691-702.

Bergstrom DM et al. 2015. Rapid collapse of a sub-Antarctic alpine ecosystem: the role of climate and pathogens. Journal of Applied Ecology **52**:774–783. Available from https://doi.org/10.1111/1365-2664.12436.

Bhend J, Bathols J, Hennessy K. 2012. Climate change impacts on snow in Victoria.

Billings WD, Bliss LC. 1959. An Alpine Snowbank Environment and Its Effects on Vegetation, Plant Development, and Productivity. Ecology **40**:388–397.

Bland LM, Keith DA, Miller RM, Murray NJ, Rodríguez JP. 2017. Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria Version 1.1. IUCN International Union for Conservation of Nature, Gland, Switzerland. Available from https://portals.iucn.org/library/sites/library/files/documents/2016-010-v1.1.pdf.

Bliss A, Pryor L, Bowman D. 2021. Lack of reliable post-fire recovery mechanisms makes the iconic Tasmanian conifer Athrotaxis cupressoides susceptible to population decline. Australian Journal of Botany **69**:162–173.

BOM. 2021. El Niño Southern Oscillation (ENSO). Available from http://www.bom.gov.au/climate/about/australian-climate-influences.shtml?bookmark=enso#:~:text=This occurs every three to,Niña%2C or opposite%2C phase.

BOM. 2025. Water Data Online. Commonwealth of Australia, Bureau of Meteorology. Available from http://www.bom.gov.au/waterdata/.

BOM, CSIRO. 2018. State of the Climate 2018. Canberra.

BOM, CSIRO. 2020. State of the Climate 2020. Available from http://www.bom.gov.au/state-of-the-climate/documents/State-of-the-Climate-2020.pdf.

Bond N. 2022. hydrostats - Hydrologic Indices for Daily Time Series Data. Available from https://cran.r-project.org/web/packages/hydrostats/hydrostats.pdf.

Boulton A, Brock M, Robson B, Ryder D, Chambers J, Davis J. 2014. Australian freshwater ecology: processes and management. John Wiley & Sons.

Boulton AJ, Daltry T, Kasahara T, Mutz M, Stanford JA. 2010. Ecology and management of the hyporheic zone: stream-groundwater interactions of running waters and their floodplains. Journal of the North American Benthological Society **29**:26–40.

Boulton AJ, M. A. Brock. 1999. Australian freshwater ecology – Processes and Management. Gleneagles Publishing, Glen Osmond, South Australia.

Bowman DM, Bliss A, Bowman CJ, Prior LD. 2019. Fire caused demographic attrition of the Tasmanian palaeoendemic conifer Athrotaxis cupressoides. Austral Ecology **44**:1322–1339.

Bowman DMJS, Williamson GJ, Gibson RK, Bradstock RA, Keenan RJ. 2021. The severity and extent of the Australia 2019–20 Eucalyptus forest fires are not the legacy of forest management. Nature Ecology & Evolution **5**:1003–1010.

Bowman W. 1992. Inputs and storage of nitrogen in winter snowpack in an alpine ecosystem. Arctic, Antarctic and Alpine Research **24**:211–215.

Bradstock R, Penman T, Boer M, Price O, Clarke H. 2014. Divergent responses of fire to recent warming and drying across south-eastern Australia. Global Change Biology **20**:1412–1428.

Brereton R, Bennett S, Mansergh I. 1995. Enhanced greenhouse climate change and its potential effect on selected fauna of SE Australia: a trend analysis. Biological Conservation **72**:339–54.

Bridle K, Kirkpatrick J. 2003. Vegetation change related to climate change in Australian subalpine environments. Pages 33–34 Climate impacts on biodiversity in Australia.

Bridle K, Kirkpatrick J, Gibson N, Balmer J. 2017. Clemes Tarn Snow Patch , Mt Field , Southwest Tasmania.

Bridle K, Kirkpatrick JB. 1997. Local environmental correlates of variability in the organic soils of moorland and alpine vegetation, Mt Sprent, Tasmania. Australian Journal of Ecology **22**:196–205.

Bridle KL, Kirkpatrick JB. 1999. Comparative Effects of Stock and Wild Vertebrate Herbivore Grazing on Treeless Subalpine Vegetation, Eastern Central Plateau, Tasmania. Autralian Journal of Botany **47**:165–188.

Brighenti S, Tolotti M, Bruno MC, Wharton G, Pusch MT, Bertoldi W. 2019. Ecosystem shifts in Alpine streams under glacier retreat and rock glacier thaw: A review. Science of The Total Environment **675**:542–559.

Brittain JE. 1982. The biology of mayflies. Annual Review of Entomology **27**:119–147.

Brittain JE. 1991. Effect of temperature on egg development in the Australian stonefly genus, Austrocercella Illies (Plecoptera: Notonemouridae). Australian Journal of Marine and Freshwater Research **42**:107–114.

Brittain JE. 1997. Egg development in Thaumatoperla, an endangered stonefly genus, endemic to the Australian Alps (Plecoptera: Eustheniidae). Pages 30–33 in P. Landolt and M. Sartori, editors. Ephemeroptera & Plecoptera: Biology-Ecology-Systematics. MTL, Fribourg.

Brittain JE. 2008. Mayflies, biodiversity and climate change. Pages 1–14 in F. R. Hauer, J. A. Stanford, and R. L. Newell, editors. International advances in the ecology, zoogeography and systematics of mayflies and stoneflies. University of California Publications in Entomology.

Brittain JE, Campbell IC. 1991. The effect of temperature on egg development in the Australian mayfly genus Coloburiscoides (Ephemeroptera: Coloburiscidae) and its relationship to distribution and life history. Journal of Biogeography **18**:231–235.

Broome L, Ford F, Dawson M, Green K, Little D, McElhinney N. 2013. Re-assessment of Mountain Pygmy-possum Burramys parvus population size and distribution of habitat in Kosciuszko National Park. Australian Zoologist **36**:381–403.

Brown LE, Hannah DM, Milner AM. 2007. Vulnerability of alpine stream biodiversity to shrinking glaciers and snowpacks. Global Change Biology **13**:958–966. Available from https://doi.org/10.1111/j.1365-2486.2007.01341.x.

Buckney RT. 1980. Chemistry of Australian waters: the basic pattern, with comments on some ecological implications. Pages 12–22 in W. D. Williams, editor. An Ecological Basis for Water Resources Management. ANU Press, Canberra.

Buckney RT, Tyler PA. 1973. Chemistry of Tasmanian inland waters. Internationale Revue der gesamten Hydrobiologie **58**:61–78.

Butt MA, Zafar M, Ahmed M, Shaheen S, Sultana S. 2021. Types of wetland and wetland plants. Pages 35–54 Wetland plants: a source of nutrition and ethno-medicines.

Cairns S. 2019. Feral horses in the Australian Alps: the analysis of aerial surveys conducted in April-May, 2014 and April-May 2019:56.

Cairns S. 2020. The results of a survey of the wild horse populations in the Kosciuszko National Park, October-November, 2020. Armidale.

Cairns S, Robertson G. 2015. 2014 Survey of Feral Horses (Equus ferus caballus) in the Australian Alps. Available from https://theaustralianalpsnationalparks.org/wp-content/uploads/2016/05/2014-aerial-survey-feral-horses.pdf.

Camac JS. 2014. Climate change and trajectories of vegetation change in an alpine heathland.

Camac JS et al. 2021. Predicting species and community responses to global change using structured expert judgement: An Australian mountain ecosystems case study. Global Change Biology **27**:1–34. John Wiley and Sons Inc. Available from https://doi.org/10.1101/2020.09.23.309377.

Camac JS, Williams RJ, Wahren C-HH, Jarrad F, Hoffmann AA, Vesk PA. 2015. Modeling rates of life form cover change in burned and unburned alpine heathland subject to experimental warming. Oecologia **178**:615–628.

Camac JS, Williams RJ, Wahren CH, Hoffmann AA, Vesk PA. 2017. Climatic warming strengthens a positive feedback between alpine shrubs and fire. Global Change Biology **23**:3249–3258. Blackwell Publishing Ltd.

Camac JS, Williams RJ, Wahren CH, Morris WK, Morgan JW. 2013. Post-fire regeneration in alpine heathland: Does fire severity matter? Austral Ecology **38**:199–207.

Campbell IC, McKaige ME, Lake. PS. 1986. The fauna of Australian high mountain streams: ecology, zoogeography and evolution. Flora and fauna of alpine Australasia: Ages and origins **83**.

Canadell JG, Meyer CP (Mick., Cook GD, Dowdy A, Briggs PR, Knauer J, Pepler A, Haverd V. 2021. Multi-decadal increase of forest burned area in Australia is linked to climate change. Nature Communications **12**. Springer US.

Carroll R, Wright IA, Reynolds JK. 2023. Loss of soil carbon in a world heritage peatland following a bushfire. International Journal of Wildland Fire **32**:1059–1070. Available from https://doi.org/10.1071/WF22204.

CGIAR-CSI. 2018. SRTM 90m DEM Digital Elevation Database. CGIAR - Consortium for Spatial Information. Available from https://srtm.csi.cgiar.org.

Chang JC, Woodward C, Shulmeister J. 2014. A snapshot of the limnology of eastern Australian water bodies spanning the tropics to Tasmania: the land-use, climate, limnology nexus. Marine and Freshwater Research **65**:872–883.

Chen J, Chang H. 2022. A review of wildfire impacts on stream temperature and turbidity across scales. Progress in Physical Geography: Earth and Environment **47**:369–394. Available from https://doi.org/10.1177/03091333221118363%0A.

Cherubin RC, Venn SE, Driscoll DA, Doherty TS, Ritchie EG. 2019. Feral horse impacts on threatened plants and animals in sub-alpine and montane environments in Victoria, Australia. Ecological Management and Restoration **20**:47–56. Blackwell Publishing.

Chessman BC. 2003. New sensitivity grades for Australian river macroinvertebrates. Marine and Freshwater Research, **54**:95–103.

Christianson KR, Johnson BM, Hooten MB. 2019. Compound effects of water clarity, inflow, wind and climate warming on mountain lake thermal regimes. Aquatic Sciences **82**.

Čiamporová-Zat’ovičová Z. 2011. Aquatic insects of remote Alpine lakes: Taxonomic composition, ecological patterns and seasonal dynamics in altitudinal environmental gradient (Tatra Mts, Slovakia). Pages 1–46 in J. G. Schmidt, editor. Alpine environment: geology, ecology, and conservation. Nova Science Publishers.

Claridge AW. 2016. Introduced Deer: Field Identification Guide For The Australian Alps. Office of Environment and Heritage, NSW National Parks and Wildlife Service, Queanbeyan, New South Wales.

Clarke H, Evans JP. 2019. Exploring the future change space for fire weather in southeast Australia. Theoretical and Applied Climatology **136**:513–527. Theoretical and Applied Climatology.

Clarke HG, Smith PL, Pitman AJ. 2011. Regional Signatures of Future Fire Weather over Eastern Australia from Global Climate Models. International Journal of Wildland Fire **20**:550. Available from https://doi.org/10.1071/WF10070.

Clarke M. 2021. The sudden death of the snow gums. ABC News Article [Online]. Available from https://www.abc.net.au/news/2021-03-10/sudden-death-of-snow-gums-longicorn-beetle/13226128 (accessed May 18, 2023).

Clarke PJ, Keith DA, Vincent BE, Letten AD. 2015. Post-grazing and post-fire vegetation dynamics: Long-term changes in mountain bogs reveal community resilience. Journal of Vegetation Science **26**:278–290. Wiley-Blackwell.

Clarke PJ, Martin ARH. 1999. Sphagnum peatlands of Kosciuszko National Park in relation to altitude, time and disturbance. Australian Journal of Botany **47**:519–536.

Clemann N, Hunter D, Scroggie M, Pietsch R, Hollis G. 2009. Vanishing frogs: prevalence of the Amphibian Chytrid Fungus (Batrachochytrium dendrobatidis) in populations of key species in the Australian Alps. Heidelberg, Victoria.

Clements AR, Suter PJ, Fussell M, Silvester E. 2016. Macroinvertebrate communities in spring-fed alpine source pools. Hydrobiologia **777**:119–138.

Closs G, Downes B, Boulton A. 2009. Freshwater Ecology: A Scientific Introduction. Wiley-Blackwell.

Coates F, Cullen PJ, Zimmer H, Shannon J. 2012. How snow gum forests and sub-alpine peatlands recover after fire: Black Saturday Victoria 2009 – Natural values fire recovery program. Heidelberg, Victoria.

Collins L, Clarke H, Clarke MF, Gausden SCM, Nolan RH, Penman T, Bradstock R. 2022. Warmer and drier conditions have increased the potential for large and severe fire seasons across south-eastern Australia. Global Ecology and Biogeography **31**:1933–1948. Available from https://doi.org/10.1111/geb.13514.

Costin A., Gray M, Totterdell C, Wimbush D. 2000. Kosciuszko Alpine FloraSecond edn. CSIRO Publishing, Collingwood, Victoria.

Costin AB. 1952. Hydrological studies in the Upper Snowy Catchment Area with special reference to the effects of land utilization. Journal of Soil Conservation Service NSW **8**:5–16.

Costin AB. 1954. A study of the ecosystems of the Monaro region of New South Wales, with special reference to soil erosion. AH Pettifer, NSW Government, Sydney.

Costin AB. 1957. High Mountain Catchments in Victoria in Relation to Land Use. Soil Conservation Authority of Victoria.

Costin AB. 1967. Alpine ecosystems of the Australasian region. Pages 57–87. in H. Wright and W. Osburn, editors. Arctic and Alpine Environments. Methuen, London.

Costin AB, Gray M, Totterdell CJ, Wimbush DJ. 1979. Kosciuszko Alpine Flora. CSIRO, Melbourne.

Costin AB, Wimbush D. 1972. Kosciusko Primitive Area: Vegetation and associated features. Map produced by CSIRO Division of Plant Industry from alpine and subalpine areas mapped by Dane Wimbush in 1966.

Cremer KW. 1995. Willows spreading by seed - implications for Australian river management. Australian Journal of Soil and Water Conservation **8**:18–27.

Crossman S. 2025. Digital Elevation Model (DEM) of Australia with 1 Second Grid WMS. Geoscience Australia, Canberra. Available from https://ecat.ga.gov.au/geonetwork/srv/eng/catalog.search#/metadata/89718.

Crossman S, Li O. 2015. Surface Hydrology Polygons (National). Geoscience Australia. Available from https://ecat.ga.gov.au/geonetwork/srv/eng/catalog.search#/metadata/83135.

CSIRO. 1972. Kosciuszko National Park Alpine Vegetation 1966 VIS_ID 4842. Department of Planning, Industry and Environment. Available from http://data.nsw.gov.au/data/dataset/kosciuszko-national-park-alpine-vegetation-1966-vis_id-4842, https://researchdata.ands.org.au/kosciuszko-national-park-visid-4842.

Cullen PJ, Kirkpatrick JB. 1988. The ecology of Athrotaxis D. Don (Taxodiaceae). I. Stand structure and regeneration of A. cupressoides. Australian Journal of Botany **36**:547–560.

Dahm CN, Candelaria-Ley RI, Reale CS, Reale JK, Van Horn DJ. 2015. Extreme water quality degradation following a catastrophic forest fire. Freshwater Biology **60**:2584–2599.

Davis CJ. 2013. Towards the development of long-term winter records for the Snowy Mountains. Australian Meteorological and Oceanographic Journal **63**:303–314.

DAWE. 2020. Australia's RAMSAR Sites. Available from https://www.environment.gov.au/water/wetlands/publications/factsheet-australias-ramsar-sites.

Dawson MJ, Miller C. 2008. Aerial mark–recapture estimates of wild horses using natural markings. Wildlife Research **35**:365–70.

DCCEEW. 2003a. Kosciuszko National Park Resorts Vegetation Thredbo 2003 Vis_ID 4841. Available from https://datasets.seed.nsw.gov.au/dataset/kosciuszko-national-park-resorts-vegetation-thredbo-2003-vis_id-4841.

DCCEEW. 2003b. Kosciuszko National Park Resorts Vegetation Perisher 2003 VIS_ID 4840. Available from https://datasets.seed.nsw.gov.au/dataset/kosciuszko-national-park-resorts-vegetation-perisher-2003-vis_id-4840.

DCCEEW. 2003c. Kosciuszko National Park Resorts Vegetation Mount Selwyn 2003 VIS_ID 4839. Available from https://datasets.seed.nsw.gov.au/dataset/kosciuszko-national-park-resorts-vegetation-mount-selwyn-2003-vis_id-4839.

DCCEEW. 2003d. Kosciuszko National Park Resorts Vegetation Charlotte Pass 2003 VIS_ID 4838. Available from https://datasets.seed.nsw.gov.au/dataset/kosciuszko-national-park-resorts-vegetation-charlotte-pass-2003-vis_id-4838.

DCCEEW. 2003e. Kosciuszko National Park Resorts Vegetation Bullocks Flat 2003 VIS_ID 4837. Available from https://datasets.seed.nsw.gov.au/dataset/kosciuszko-national-park-resorts-vegetation-bullocks-flat-2003-vis_id-4837.

DCCEEW. 2011a. Forest Ecosystems: Vegetation of the Southern Forests. VIS ID 3858. State Government of NSW and NSW Department of Climate Change, Energy, the Environment and Water. Available from https://datasets.seed.nsw.gov.au/dataset/c13950e1-9afd-4aa6-8064-8783f3d4fd57.

DCCEEW. 2011b. Forest Ecosystems: Native Vegetation of the Southern Forests: South-east Highlands, Australian Alps, South-west Slopes, and SE Corner bioregions. Pre-1750. VIS ID 3859. Available from https://datasets.seed.nsw.gov.au/dataset/forest-ecosystems-native-vegetation-of-the-southern-forests-south-east-highlands-australian-38a92.

De Deckker P, Hancock GJ, Olley JM, Stanley S, Hope G. 2023. The effect of the introduction of livestock on the erosion of alpine soils: a comparison of five dating techniques applied to sediments of the Australian alpine Blue Lake. Journal of Paleolimnology **70**:77–93.

Dean JC, Cartwright DI. 1992. Plecoptera, Ephemeroptera and Trichoptera of the Pelion Valley, Tasmanian World Heritage Area. Occasional Papers from the Museum of Victoria **5**:73–79.

DeBano LF. 2003. The role of fire and soil heating on water repellency. Soil Water Repellency: Occurrence, Consequences, and Amelioration **232**:193–202.

DECC. 2008. Ecological character description: Blue Lake Ramsar site. Available from https://www.environment.nsw.gov.au/-/media/OEH/Corporate-Site/Documents/Water/Wetlands/ecological-character-description-blue-lake-ramsar-site-080273.pdf (accessed April 24, 2020).

DELWP. 2015. Flora and Fauna Gaurantee Action Statement No. 266. Soil erosion and vegetation damage and disturbance in the alpine regions of Victoria caused by cattle grzing. Melbourne.

DELWP. 2016. National Recovery Plan for the Mountain Pygmy-possum Burramys parvus. Canberra. Available from https://www.environment.gov.au/system/files/resources/14b32262-159e-4462-97fe-f13152aaa461/files/national-recovery-plan-mountain-pygmy-possum.pdf.

DELWP. 2018a. Native Vegetation - Modelled Extent 2005. Available from https://datashare.maps.vic.gov.au/search?q=uuid%3D58e3ca32-e951-5bfa-9974-728cea56a14e.

DELWP. 2018b. Native Vegetation - Modelled 1750 Ecological Vegetation Classes. Available from https://datashare.maps.vic.gov.au/search?q=uuid%3D853bda8a-a680-5ec4-9cbe-8a33b51eee2b.

DELWP. 2021. Native Vegetation – Modelled Extent 2005. Available from https://discover.data.vic.gov.au/dataset/native-vegetation-modelled-extent-2005 (accessed February 4, 2021).

Di Luca A, Evans JP, F. Ji. 2018. Australian snowpack in the NARCliM ensemble: evaluation, bias correction and future projections. Climate Dynamics **51**:639–666.

Dixon KM, Cary GJ, Renton M, Worboys GL, Gibbons P. 2019. More long-unburnt forest will benefit mammals in Australian sub-alpine forests and woodlands. Austral Ecology **44**:1150–1162.

Dodson JR. 2001. A vegetation and fire history in a subalpine woodland and rain-forest region, Solomons Jewel Lake, Tasmania. The Holocene **11**:111–116.

Dolezal J, Kurnotova M, Stastna P, Klimesova J. 2020. Alpine plant growth and reproduction dynamics in a warmer world. New Phytologist **228**:1295–1305.

DOTE. 2015. National recovery plan for the Alpine Sphagnum bogs and associated fens community.

Dowdy AJ, Ye H, Pepler A, Thatcher M, Osbrough SL, Evans JP, Di Virgilio G, McCarthy N. 2019. Future changes in extreme weather and pyroconvection risk factors for Australian wildfires. Scientific Reports **9**:1–11. Nature Publishing Group. Available from http://dx.doi.org/10.1038/s41598-019-46362-x.

DPI. 2023. Distribution maps for vertebrate pests. Available from https://www.dpi.nsw.gov.au/biosecurity/vertebrate-pests/publications/distribution-maps-for-vertebrate-pests (accessed May 19, 2023).

DPIE. 2020. Blue Lake. Available from https://www.environment.nsw.gov.au/topics/water/wetlands/internationally-significant-wetlands/blue-lake.

DPIE. 2021. Snow Gum dieback mapping in Kosciuszko National Park Thredbo, Perisher, Charlottes resorts. Available from https://datasets.seed.nsw.gov.au/dataset/snow-gum-dieback-mapping-kosciuszko-national-park-resorts-2021.

DPIPWE. 2010. Vulnerability of Tasmania's Natural Environment to Climate Change: An overview. Available from http://dpipwe.tas.gov.au/Documents/Vulnerability-of-Tasmanias-Natural-Environment-to-Climate-Change---An-Overview.pdf.

DPIPWE. 2016a. Department of Primary Industries, Parks, Water and Environment (DPIPWE) Submission Legislative Council Goverment Administration Committee A: Inquiry into Wild Fallow Deer.

DPIPWE. 2016b. Lake Mackenzie Alpine Fire Impacts Workshop, 8 June 2016. Natural and Cultural

Heritage Division, Department of Primary Industries Parks Water and Environment, Hobart, Nature Conservation Report Series 16/2.

DPIPWE. 2018. Tasmanian Threatened Native Vegetation Communities - Sphagnum Peatland. Tasmania.

DPIPWE. 2020. TASVEG 4.0, Released July 2020. Tasmanian Vegetation Monitoring and Mapping Program, Natural and Cultural Heritage Division. Available from https://dpipwe.tas.gov.au/conservation/development-planning-conservation-assessment/planning-tools/monitoring-and-mapping-tasmanias-vegetation-(tasveg)/tasveg-the-digital-vegetation-map-of-tasmania (accessed February 4, 2021).

Driessen MM, Bonham K, Balmer J. 2023. Invertebrate responses to bushfire in highland grassy sedgeland (1-5 years). Nature Conservation Report 23/3. Hobart.

Driessen MM, Mallick SA. 2003. The vertebrate fauna of the Tasmanian Wilderness World Heritage Area. Pacific Conservation Biology **9**:187–206.

Driscoll DA et al. 2019. Impacts of feral horses in the Australian Alps and evidence-based solutions. Ecological Management and Restoration **20**:63–72. Blackwell Publishing.

Drost AM, Siebers AR, Paillex A, Misteli B, Peeters ETHM, Robinson CT. 2022. Long-lasting effects of experimental flow intermittency on alpine stream macroinvertebrates (Val Roseg, Switzerland). Hydrobiologia **849**:1931–1946.

DSE. 2004. EVC/Bioregion Benchmark for Vegetation Quality Assessmen. Department of Sustainability and Environment. Available from https://www.environment.vic.gov.au/biodiversity/bioregions-and-evc-benchmarks.

DSE. 2013. Advisory List of Threatened Vertebrate Fauna in Victoria. Melbourne.

DSE. 2020a. Bioregional conservation status for each BioEVC.

DSE. 2020b. Victorian Fires - Fire Extent 2019/2020. State of Victoria. Available from https://www.ffm.vic.gov.au/__data/assets/pdf_file/0022/500728/Victorian_fires_fire_area_end1920_Victorian-Fires.pdf.

Duretto M. 2018. Notice of and reasons for the Final Determination. Page Ecological Management and Restoration. NSW Threatened Species Scientific Committee.

Duursma DE, Gallagher R V., Roger E, Hughes L, Downey PO, Leishman MR. 2013. Next-generation invaders? Hotspots for naturalised sleeper weeds in Australia under future climates. PLoS ONE **8**:e84222.

Ecology Australia. 2003. Kosciuszko Resorts Vegetation Assessment an unpublished report to NSW Planning. Fairfield, Victoria.

Edmonds T, Lunt ID, Roshier DA, Louis J. 2006. Annual variation in the distribution of summer snowdrifts in the Kosciuszko alpine area, Australia, and its effect on the composition and structure of alpine vegetation. Austral Ecology **31**:837–848.

Edwards ED, McQuillan PB. 1998. Lepidoptera. Pages 67–73 in M. M. Driessen, M. D. Comfort, J. Jackson, J. Balmer, A. M. M. Richardson, and P. B. McQuillan, editors. Wilderness Ecosystems Baseline Studies 1990–1993. WEBS, Pelion Plains – Mt Ossa,Parks and Wildlife Service,Hobart.

Eldridge DJ., Travers SK., Val J., Zaja A., Veblen KE. 2019. Horse Activity is Associated with Degraded Subalpine Grassland Structure and Reduced Habitat for a Threatened Rodent. Rangeland Ecology and Management **72**:467–473. Available from https://doi.org/10.1016/ j.rama.2018.12.008.

Elmendorf SC et al. 2012. Global assessment of experimental climate warming on tundra vegetation: Heterogeneity over space and time. Ecology Letters **15**:164–175.

EMM Consulting. 2018. Exploratory works for Snowy 2.0: Environmental Impact Statement (Main Report). Report prepared for Snowy Hydro. Available from https://www.snowyhydro.com.au/our-scheme/snowy20/approvals/eis/.

Enright NJ, Fontaine JB, Bowman DMJS, Bradstock RA, Williams RJ. 2015. Interval squeeze: Altered fire regimes and demographic responses interact to threaten woody species persistence as climate changes. Frontiers in Ecology and the Environment **13**:265–272.

Environment and Communications Reference Committee. 2023. Chapter 5 - Management of Feral Horses in the Australian Alps. Page Impacts and Management of Feral Horses in the Australian Alps, 169. Commonwealth of Australia. Available from https://parlinfo.aph.gov.au/parlInfo/download/committees/reportsen/RB000047/toc_pdf/ImpactsandmanagementofferalhorsesintheAustralianAlps.pdf.

EPSD. 2019. Conservation Advice - High Country Bogs and Associated Fens Ecological Community.

Evans JP, Ji F, Lee C, Smith P, Argüeso D, Fita L. 2014. Design of a regional climate modelling projection ensemble experiment - NARCliM. Geoscientific Model Development:621– 629.

Fairman TA, Bennett LT, Tupper S, Nitschke CR. 2017. Frequent wildfires erode tree persistence and alter stand structure and initial composition of a fire-tolerant sub-alpine forest. Journal of Vegetation Science **28**:1151–1165.

Fensham RJ, Kirkpatrick JB. 1992. The eucalypt forest grassland/grassy woodland boundary in central Tasmania. Australian Journal of Botany **40**:123–138.

Fick SE, Hijmans RJ. 2017. WorldClim 2: New 1-km Spatial Resolution Climate Surfaces for Global Land Areas. International Journal of Climatology **37**:4302–15. Available from https://doi.org/10.1002/joc.5086.

Filoso S, Bezerra MO, Weiss KCB, Palmer MA. 2017. Impacts of forest restoration on water yield: A systematic review. PLoS ONE **12**:e0183210.

Finkele K, Mills GA, Beard G, Jones DA. 2006. National gridded drought factors and comparison of two soil moisture deficitformulations used in prediction of Forest Fire Danger Index in Australia. Australian Meteorological Magazine **55**:183–197.

Fire Information Systems Group. 2008. January 1939 bushfires. Department of Sustainability and Environment. Available from https://www.ffm.vic.gov.au/history-and-incidents/past-bushfires/past-bushfire-maps.

Fitzgerald NB. 2011. Establishment Report for Tasmanian Wilderness World Heritage Area Climate Change Monitoring Program: Montane Conifers (Nature Conservation Report 2011/06; Nature Conservation Report).

Fletcher MS, Wolfe BB, Whitlock C, Pompeani DP, Heijnis H, Haberle SG, Gadd PS, Bowman DMJS. 2014. The legacy of mid-holocene fire on a Tasmanian montane landscape. Journal of Biogeography **41**:476–488.

Forest Fire Management Victoria. 2023. Past Bushfires: A Chronology of Major Bushfires in Victoria from 2020 Back to 1851. Available from https://www.ffm.vic.gov.au/history-and-incidents/past-bushfires.

Forsyth SM. 2003. Geology of the Mount Koonya area. Tasmanian Geological Survey Record **8**:1–31.

Foulkes J, Prior L, Leonard S, Bowman D. 2021. Demographic effects of severe fire in montane shrubland on Tasmania’s Central Plateau. Fire **4**:32.

Fraser IP, Williams RJ, Murphy BP, Camac JS, Vesk PA. 2016. Fuels and landscape flammability in an Australian alpine environment. Austral Ecology **41**:657–670.

FSC. 2011. Final Determination: Aquatic Ecological Community in the Catchment of the Snowy River in NSW. Fisheries Scientific Committee. Available from https://www.dpi.nsw.gov.au/__data/assets/pdf_file/0005/636539/Snowy-River-EEC-Final-Determination.pdf.

Fulton W, Tyler P. 1993. Fauna and flora of the lakes and tarns. Pages 109–113 in S. Smith and M. Banks, editors. Tasmanian Wilderness – World Heritage Values. Royal Society of Tasmania, Hobart.

Fureder L, Schutz C, Wallinger M, Burger R. 2001. Physico-chemistry and aquatic insects of a glacier-fed and a spring-fed alpine stream. Freshwater Biology **46**:1673–1690.

Gabbud C, Lane SN. 2016. Ecosystem impacts of Alpine water intakes for hydropower: the challenge of sediment management. Wires Water **3**:1.

Gallant J, Wilson N, Dowling T, Read A, Inskeep C. 2019. SRTM-derived 1 Second Digital Elevation Models Version 1.0. Available from https://ecat.ga.gov.au/geonetwork/srv/eng/catalog.search#/metadata/72759.

Gellie NJH. 2005. Native Vegetation of the Southern Forests: South-east Highlands, Australian Alps, South-west Slopes, and SE Corner bioregions. Cunninghamia **9**:219–254.

Geoscience Australia. 2021. National Surface Water Information. Available from https://www.ga.gov.au/scientific-topics/national-location-information/national-surface-water-information.

Gibbons F, Rowan J. 1993. Soils in relation to vegetation in Victoria. Pages 159–194 in D. B. Foreman and N. G. Walsh, editors. Flora of Victoria, Volume 1 - Introduction. Inkata Press, Melbourne.

Gibson N. 1984. Impact of trampling on bolster heath communities of Mount Field National Park. Papers and Proceedings of the Royal Society of Tasmania **118**:47–52.

Gibson N, Barker PC, Cullen P, Shapcott A. 1995. Conifers of southern Australia. In Ecology of the Southern Conifer. Melbourne University Press.

Gibson N, Kirkpatrick J. 1985. Vegetation and flora associated with localized snow accumulation at Mount Field West, Tasmania. Australian Journal of Ecology, **10**:91–99.

Giglio L, Descloitres J, Justice CO, Kaufman Y. 2003. An enhanced contextual fire detection algorithm for MODIS. Remote Sensing of Environment **87**:273–282. Available from http://dx.doi.org/10.1016/S0034-4257(03)00184-6.

Gimenez A, Pastor E, Zárate L, Planas E, Arnaldos J. 2004. Long-term forest fire retardants: a review of quality, effectiveness, application and environmental considerations. International Journal of Wildland Fire, **13**:1–15.

Godfree R, Lepschi B, Mallinson D. 2004. Ecological filtering of exotic plants in an Australian sub-alpine environment. Journal of Vegetation Science **15**:227–36.

González-Orozco CE et al. 2016. Phylogenetic approaches reveal biodiversity threats under climate change. Nature Climate Change **6**:1110–1114. Available from https://doi.org/10.1038/nclimate3126.

Good R. 1982. The effects of prescribed burning in the subalpine area of Kosciuszko National Park. University of NSW.

Good R. 1992. Kosciusko heritage : the conservation significance of Kosciusko National Park. National Parks and Wildlife Service, New South Wales.

Good R. 1998. Changing snow regimes and the distribution of alpine vegetation. Pages 98–112 in K. Green, editor. Snow a natural history; an uncertain future. Australian Alps Liaison Committee, Surrey Beatty & Sons Pty. Ltd., Canberra.

Good R. 2006. The Australian Alps Rehabilitation manual.

Good R, Wright G, Whinam J, Hope G. 2010. Restoration of mires of the Australian Alps following the 2003 wildfires. Page in S. G. Haberle, J. Stevenson, and M. Prebble, editors. Altered Ecologies: Fire, climate and human influence on terrestrial landscapes. ANU Press.

Green K. 1988. A Study of Antechinus swainsonii and Antechinus stuartii and their prey in the Snowy Mountains. Australian National University.

Green K, editor. 2002. Biodiversity in the Snowy Mountains. Australian Institute of Alpine Studies, Jindabyne, NSW.

Green K. 2008. Fragmented distribution of a rock climbing fish, the Mountain Galaxias Galaxias olidus, in the snowy mountains. Proceedings of the Linnean Society of New South Wales **129**:175–182.

Green K. 2010. Alpine taxa exhibit differing responses to climate warming in the Snowy Mountains of Australia. Journal of Mountain Science **7**:167–175.

Green K. 2011. Interannual and seasonal changes in the ice cover of glacial lakes in the Snowy Mountain of Australia. Journal of Mountain Science **8**:655–663.

Green K. 2012. Intra- and inter-annual changes in chemistry of Australian glacial lakes. Marine and Freshwater Research **63**:513–527.

Green K. 2014. Alpine ecosystems. Pages 91–98 in D. B. Lindenmayer, S. Dovers, and S. Morton, editors. Ten Commitments Revisited: Securing Australia's Future Environment.

Green K. 2016. Adaptations of large marsupials to survival in winter snow cover: Locomotion and foraging. Canadian Journal of Zoology **94**:753–759. Canadian Science Publishing.

Green K, Osborne W. 1994. Wildlife of the Australian snow-country, 1st edition. Reed Books, Chatswood, Sydney, Sydney.

Green K, Osborne W. 2012. Field Guide to Wildlife of the Australian Snow-country. Reed New Holland Publishers, Sydney.

Green K, Pickering C. 2009a. The decline of snowpatches in the snowy mountains of australia: Importance of climate warming, variable snow, and wind. Arctic, Antarctic, and Alpine Research **41**:212–218.

Green K, Pickering C. 2009b. Vegetation, microclimate and soils associated with the latest-lying snowpatches in Australia. Plant Ecology and Diversity **2**:289–300.

Green K, Pickering C. 2013. Limited Effect of Hare Grazing and Short-Term Climatic Variations on the Most Common Alpine Vegetation Community in the Snowy Mountains, Australia. Plant Ecology and Diversity **6**:511–22.

Green K, Pickering CM. 2002. A scenario for mammal and bird diversity in the Snowy Mountains of Australia in relation to climate change. Page in C. Korner and E. M. Spehn, editors. Mountain Biodiversity: A Global Assessment. Parthenon, London.

Green K, Sanecki G. 2006. Immediate and short-term responses of bird and mammal assemblages to a subalpine wildfire in the Snowy Mountains, Australia. Austral Ecology **31**:673–681.

Green K, Stein JA. 2015. Modeling the Thermal Zones and Biodiversity on the High Mountains of Meganesia: The Importance of Local Differences. Arctic, Antarctic, and Alpine Research **47**:671–680.

Griffin PC, Hoffmann AA. 2012. Mortality of Australian alpine grasses (Poa spp.) after drought: Species differences and ecological patterns. Journal of Plant Ecology **5**:121–133.

Grose M, Barnes-Keoghan I, Corney S, White C, Holz G, Bennett J, Gaynor S, Bindoff N. 2010. Climate Futures for Tasmania: general climate impacts technical report. Antarctic Climate & Ecosystems Cooperative Research Centre, Hobart, Tasmania. Climate Futures for Tasmania **3**.

Grover SPP, McKenzie BM, Baldock JA, Papst WA. 2005. Chemical characterisation of bog peat and dried peat of the Australian Alps. Australian Journal of Soil Research **43**:963–971.

Gust N, Griffiths J. 2009. Platypus mucormycosis and its conservation implications. Australasian Mycologist **28**:1–8.

Hancock MA, Timms B V., Morton JK, Renshaw BA. 2000. The structure of the littoral invertebrate communities of the Kosciuszko region lakes. Proceedings of the Linnean Society of New South Wales **122**:69–77.

Hardie SA, Jackson JE, Barmuta LA, White RW. 2006. Status of galaxiid fishes in Tasmania, Australia: conservation listings, threats and management issues. Aquatic Conservation: Marine and Freshwater Ecosystems **16**:235-250.

Harding MC, Kirkpatrick JB. 2018. Short-term recovery of cushion plant communities after fire on the Central Plateau, Tasmania. Papers and Proceedings of the Royal Society of Tasmania **152**:9–16. Royal Society of Tasmania.

Harris I, Jones PD, Osborn TJ, Lister D. 2014. Updated high-resolution grids of monthly climatic observations - the CRU TS3.10 Dataset. International Journal of Climatology **34**:623–642.

Harris RMB, Remenyi T, Bindoff NL. 2016. The Potential Impacts of Climate Change on Victorian Alpine Resorts. A Report for the Alpine Resorts Co-ordinating Council.

Harris S, Kitchener A. 2005. From Forest to Fjaeldmark: Descriptions of Tasmania's Vegetation. Department of Primary Industries, Water and Environment, Printing Authority of Tasmania, Hobart.

Harrison-Day V, Annandale B, Balmer J, Kirkpatrick JB. 2016. Decadal-scale vegetation dynamics above the alpine treeline, Mount Rufus, Tasmania. Papers and Proceedings of the Royal Society of Tasmania **150**:9–18. Royal Society of Tasmania.

Hart BT, McKelvie ID. 1985. Chemical Limnology in Australia. Pages 3–31 in P. De Deckker and W. D. Williams, editors. Limnology in Australia. CSIRO Australia; Dr W. Junk Publishers, Melbourne, Dorderecht.

Hartley R, Blanchard W, Schroder M, Lindenmayer DB, Sato C, Scheele BC. 2022. Exotic Herbivores Dominate Australian High-elevation Grasslands. Conservation Science and Practice **4**:e601. Available from https://doi.org/10.1111/csp2.601.

Hartmann H et al. 2018. Research frontiers for improving our understanding of drought-induced tree and forest mortality. New Phytologist **218**:15–28. Available from https://doi.org/10.1111/nph.15048.

Harvey A. 2007. Fire Severity Mapping of Mountain Pygmy-possum Burramys parvus Habitat at Mount Buller, Victoria. Final Report to the Mount Buller and Mount Stirling 37 Alpine Resort Management Board. Project No. 6249. Melbourne.

Hazell D. 2003. Frog ecology in modified Australian landscapes: A review. Wildlife Research **30**:193–205.

Heinze D, Harvey A. 2006. Habitat mapping for the Mountain Pygmy-possum Burramys parvus at Mount Buller, Victoria. Report to Buller Ski Lifts and the Mount Buller and Mount Stirling Alpine Resort Management Board. Project No. 5507. Melbourne.

Hennessy K, FItzharris B, Bates B, Harvey N, Howden M, Hughes L, Salinger J, Warrick R, Becken S, Chambers L. 2007. Australia & New Zealand. Pages 507–540 Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press.

Hennessy K, Whetton P., Walsh K, Smith I., Bathols J., Hutchinson M, Sharples J. 2008. Climate Change Effects on Snow Conditions in Mainland Australia and Adaptation at Ski Resorts through Snowmaking. Climate Research **35**:255–70. Available from https://doi.org/10.3354/cr00706.

Hennessy K, Whetton P, Smith I, Bathols J, Hutchinson M, Sharples J. 2003. The impact of climate change on snow conditions in mainland Australia. Page CSIRO.

Hennessy KJ. 2007. Fire weather. Page in K. B. Pearce, P. N. Holper, M. Hopkins, W. J. Bouma, P. H. Whetton, K. J. Hennessy, and S. B. Power, editors. Climate Change in Australia: Technical Report 2007. CSIRO Marine and Atmospheric Research, Aspendale.

Hennessy KJ, Lucas C, Nicholls N, Bathols J, Suppiah R, Ricketts J. 2005. Climate change impacts on fire-weather in south-east Australia. Page CSIRO Marine and Atmospheric Research. CSIRO Marine and Atmospheric Research.

Henríquez WI, Fletcher M-S, Romano A. 2023. Vegetation, fire and climate history in central-western Tasmania (41° S), Australia, over the last~ 21,000 years. Quaternary Science Reviews **318**:108315.

Hickman IT, Camac JS, Venn SE, Morgan JW. 2024. Long-term alpine summit vegetation cover change: Divergent trajectories driven by climate warming and fire. Arctic, Antarctic, and Alpine Research **56**. Available from https://doi.org/10.1080/15230430.2024.2429864.

Hijmans RJ. 2025. terra: Spatial Data Analysis. R package version 1.8-57. Available from https://github.com/rspatial/terra.

Hines F, Tolhurst KG, Wilson AA, Mccarthy GJ. 2010. Overall fuel hazard assessment guide. In Fire and adaptive management. Victorian Government Department of Sustainability and Environment:1– 42.

Hoffmann AA, Camac JS, Williams RJ, Papst W, Jarrad FC, Wahren CH, Hoffmann AA, Camac JS, Williams RJ, Warwick P. 2010. Phenological changes in six Australian subalpine plants in response to experimental warming and year-to-year variation. Journal of Ecology **98**:927–937.

Holt G. 2025. hydrogauge: Pulls information from the Kisters Kiwis and Hydstra APIs. R package version 0.1.1.9010. Available from https://galenholt.github.io/hydrogauge/, https://github.com/galenholt/hydrogauge/.

Holz A, Wood S, Ward C, Veblen T, Bowman D. 2020. Population collapse and retreat to fire refugia of the Tasmanian endemic conifer Athrotaxis selaginoides following the transition from Aboriginal to European fire management. Global Change Biology **26**:3108–3121.

Holz AA, Wood SWSW, Veblen TTT, Bowman DMJSDMJSJS. 2015. Effects of high-severity fire drove the population collapse of the subalpine Tasmanian endemic conifer Athrotaxis cupressoides. Global Change Biology **21**:445–458. WILEY-BLACKWELL, 111 RIVER ST, HOBOKEN 07030-5774, NJ USA.

Hone J. 2002. Feral Pigs in Namadgi National Park, Australia: Dynamics, Impacts and Management. Biological Conservation **105**:231–42. Available from https://doi.org/10.1016/S0006-3207(01)00185-9.

Hope G, Nanson R, Jones P. 2012. Peat-forming bogs and fens of the Snowy Mountains of NSW: technical report. Office of Environment and Heritage, ACT, Canberra.

Hope G, Whinam J, Good R. 2005. Methods and preliminary results of post-fire experimental trials of restoration techniques in the peatlands of Namadgi (ACT) and Kosciuszko National Parks (NSW). Ecological Management and Restoration **6**:2002–2005.

Hope GS, Nanson R, Flett I, Australian National University. Department of Archaeology and Natural History. 2009. The peat-forming mires of the Australian Capital Territory. Territory and Municipal Services.

Hope GS, Nanson RA. 2015. Peatland carbon stores and fluxes in the Snowy Mountains, New South Wales, Australia. Mires and Peat **15**:Article 11, 1-23. Available from https://www.ingentaconnect.com/content/doaj/1819754x/2015/00000015/00000011/art00001%0Ahttp://mires-and-peat.net/media/map15/map_15_11.pdf.

Hope GS, Whinam J. 2005. The Peatlands of the Australasian Region. Mires - from Siberia to Tierra del Fuego **35**:397–434. Available from www.biologiezentrum.at.

Hosking J, Schroder M. 2013. Impact of Cytisus scoparius (L.) Link, broom, in naturally treeless sub-alpine frost-hollow vegetation communities areas at the Barrington Tops, New South Wales, Australia. Cunninghamia **13**:337–351.

Hughes L. 2003. Climate change and Australia: Trends, projections and impacts. Austral Ecology **28**:423–443.

Hughes L. 2011. Climate change and Australia: Key vulnerable regions. Regional Environmental Change **11**:189–195.

Hynes HBN, Hynes ME. 1975. The life histories of many of the stoneflies (Plecoptera) of south-eastern mainland Australia. Australian Journal of Marine and Freshwater Research **26**:113–153.

Ingram HAP. 1983. Hydrology. Pages 257–283 in A. J. P. Gore, editor. Ecosystems of the World (4a). Mires: Swamp, Bog, Fen and Moor. General studies. Elsevier Scientific Publishing Company, Amsterdam.

IPBES. 2019. Global assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Page (Brondízio ES, Settele J, Díaz S, Ngo HT, editors) Debating Nature's Value. IPBES secretariat, Bonn, Germany. Available from https://ipbes.net/global-assessment%0Ahttps://ipbes.net/global-assessment-report-biodiversity-ecosystem-services.

IPCC. 2022. Climate Change 2022: Impacts Adaptation and Vulnerability Summary for Policymakers. Geneva, Switzerland.

IUCN. 2024. Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria: version 2.0. Page (Keith DA et al., editors) Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria: version 2.0Version 2. IUCN, Gland, Switzerland. Available from https://portals.iucn.org/library/sites/library/files/documents/2024-021-En.pdf.

Jarman S, Kantvilas G. 1988. Lichens of buttongrass (Gymnoschoenus) moorland in Tasmania. Papers & Proceedings of the Royal Society of Tasmania **122**:1–17.

Jarrad FC, Wahren CH, Williams RJ, Burgman MA. 2008. Impacts of experimental warming and fire on phenology of subalpine open-heath species. Australian Journal of Botany **56**:617–629.

Ji F, Nishant N, Evans JP, Luca A Di, Virgilio G Di, Cheung KKW, Tam E, Beyer K, Riley. ML. 2022. Rapid Warming in the Australian Alps from Observation and NARCliM Simulations. Atmosphere **13**:1686. Available from https://doi.org/10.3390/atmos13101686.

Johnston F. 2006. Exotic Plants in the Australian Alps including a case study of the ecology of Achillea millefolium. Griffith University.

Johnston FM, Pickering CM. 2001. Alien plants in the Australian Alps. Mountain Research and Development **21**:284–291.

Jones D, Wang W, Fawcett R. 2009. High-quality spatial climate data-sets for Australia. Australian Meteorological and Oceanographic Journal **58**:233–248.

Jordan GJ, Harrison PA, Worth JRP, Williamson GJ, Kirkpatrick JB. 2016. Palaeoendemic plants provide evidence for persistence of open, well-watered vegetation since the Cretaceous. Global Ecology and Biogeography **25**. Available from https://doi.org/10.1111/geb.12389.

Jouquet P, Dauber J, Lagerlöf J, Lavelle P, Lepage M. 2006. Soil invertebrates as ecosystem engineers: intended and accidental effects on soil and feedback loops. Applied Soil Ecology **32**:153–64.

Kalabokidis KD. 2018. Effects of wildfire suppression chemicals on people and the environment. Global NEST Journal **2**:129–137.

Kalny G, Laaha G, Melcher A, Trimmel H, Weihs P, Rauch HP. 2017. The influence of riparian vegetation shading on water temperature during low flow conditions in a medium sized river. Knowledge & Management of Aquatic Ecosystems **418**:5.

Karger DN, Lange S, Hari C, Reyer CPO, Conrad O, Zimmermann NE, Frieler K. 2023. CHELSA-W5E5: Daily 1 km meteorological forcing data for climate impact studies. Earth System Science Data **15**:2445–2464. Available from https://doi.org/10.5194/essd-15-2445-2023.

Karis T, Silvester E, Rees G. 2016. Chemical regulation of alpine headwater streams during a storm event (Bogong High Plains, Victoria, Australia). Journal of Hydrology **542**:317–329. Elsevier B.V. Available from http://dx.doi.org/10.1016/j.jhydrol.2016.09.014.

Kauffman JB, Krueger WC. 1984. Livestock impacts on riparian ecosystems and streamside management implications: a review. Journal of Range Management **37**:430–8.

Kearney MR, Porter WP. 2017. NicheMapR - an R package for biophysical modelling: the microclimate model. Ecography **40**:664–674.

Keith DA. 1996. Fire-driven extinction of plant populations: a synthesis of theory and review of evidence from Australian vegetation. Proceedings-Linnean Society of New South Wales **116**:37–78.

Keith DA. 2004. Ocean shores to desert dunes, 1st edition. Department of Environment and Conservation (NSW), Hurstville, Sydney.

Keith DA et al. 2013. Scientific Foundations for an IUCN Red List of Ecosystems. PLoS ONE **8**:e62111. Available from Keith_et_al_2013_Scientific_foundations_for_an_IUCN_Red_List_of_ecosystems.pdf.

Keith DA et al. 2022a. A function-based typology for Earth's ecosystems. Nature **610**:513–518. Available from https://www.nature.com/articles/s41586-022-05318-4.

Keith DA et al. 2022b. Fire-related threats and transformational change in Australian ecosystems. Global Ecology and Biogeography **31**:2070–2084.

Keith DA, Benson DH, Baird IRC, Watts L, Simpson CC, Krogh M, Gorissen S, Ferrer-Paris JR, Mason TJ. 2023. Effects of interactions between anthropogenic stressors and recurring perturbations on ecosystem resilience and collapse. Conservation Biology **37**:1–14.

Keith DA, Ferrer-Paris JR, Nicholson E, Kingsford RT. 2020. IUCN Global Ecosystem Typology 2.0 Descriptive profiles for biomes and ecosystem functional groups. IUCN, Gland, Switzerland.

Keith DA, Pellow BJ. 2015. Review of Australia's Major Vegetation classification and descriptions. Sydney.

Kelly SR. 2009. The origin, genetic diversity and taxonomy of the invasive diatom Didymosphenia geminata (Bacilliariophyceae) in New Zealand. The University of Waikato.

Khaliq I, Hardy GESJ, McDougall KL, Burgess TI. 2019. Phytophthora Species Isolated from Alpine and

Sub-Alpine Regions of Australia, Including the Description of Two New Species; Phytophthora Cacuminis Sp. Nov and Phytophthora Oreophila Sp. Nov. Fungal Biology **123**:29–41. Available from https://doi.org/10.1016/j.funbio.2018.10.006.

Khamis K, Brown LE, Hannah DM, Milner AM. 2015. Experimental evidence that predator range expansion modifies alpine stream community structure. Freshwater Science **34**. Available from https://doi.org/10.1086/679484.

Kirkpatrick J., Hutchinson M., McQuillan P. 1993. Alpine ecosystems. Pages 73–79 in S. . Smith and M. . Banks, editors. Tasmanian Wilderness-World Heritage Values. Royal Society of Tasmania, Hobart.

Kirkpatrick J, Harwood C. 1980. The vegetation of an infrequently burned Tasmanian mountain region. Proceedings of the Royal Society of Victoria **91**:79–107.

Kirkpatrick J, Marsden-Smedley J. 2014. Snow patch and fjaeldmark map. Unpublished Digital map produced in the School of Land and Food, University of Tasmania, a copy of which is held within the electronic files of the Natural Cultural Heritage Division, Hobart.

Kirkpatrick J, Marsden-Smedley JB. 2015. Snowpatch and Feldmark map for Tasmania, Unpublished digital spatial layer DPIPWE, Hobart.

Kirkpatrick JB. 1982. Phytogeographical analysis of Tasmanian alpine floras. Journal of Biogeography **9**:255–271.

Kirkpatrick JB. 1983. Treeless plant communities of the Tasmanian high country. Proceedings of the Ecological Society of Australia **12**:61–77.

Kirkpatrick JB. 1984a. Altitudinal and successional variation in the vegetation of the northern part of the West Coast Range, Tasmania. Australian Journal of Ecology **9**:81–91.

Kirkpatrick JB. 1984b. Tasmanian high mountain vegetation II - Rocky Hill and Pyramid Mountain. Papers & Proceedings - Royal Society of Tasmania **118**:5–20.

Kirkpatrick JB. 1997. Alpine Tasmania: An Illustrated Guide to the Flora and Vegetation. Oxford University Press, Melbourne.

Kirkpatrick JB, Alvarez MA, Bridle K, Barros A. 2024. High resistance to non-native plant invasion exhibited in a maritime alpine environment. Biological Invasions **26**:4089–4103.

Kirkpatrick JB, Bridle K. 1999. Environment and floristics of ten Australian alpine vegetation formations. Australian Journal of Botany **47**:1–21.

Kirkpatrick JB, Bridle KL. 2013. Natural and cultural histories of fire differ between Tasmanian and mainland Australian alpine vegetation. Australian Journal of Botany **61**:465–474.

Kirkpatrick JB, Bridle KL. 2016. Grazing and the Absence of Fire Promote the Dominance of an Unpalatable Shrub in a Patch Mosaic Cyclic Successional System. Australian Journal of Botany **64**:45. Available from https://doi.org/10.1071/BT15162.

Kirkpatrick JB, Bridle KL, Dickinson KJM. 2010. Decades-scale vegetation change in burned and unburned alpine coniferous heath. Australian Journal of Botany **58**:453–462.

Kirkpatrick JB, Bridle KL, Lynch AJJ. 2002. Changes in alpine vegetation related to geomorphological processes and climatic change on Hill One, Southern Range, Tasmania, 1989-2000. Australian Journal of Botany **50**:753–759.

Kirkpatrick JB, Deane A, Parry J. 2019. The dynamics of rush circles in subalpine grassland. Australian Journal of Botany **67**:335–340. CSIRO.

Kirkpatrick JB, Dickinson KJM. 1984. The impact of fire on tasmanian alpine vegetation and soils. Australian Journal of Botany **32**:613–629.

Kirkpatrick JB, Gibson N. 1984. Dynamics of a Tasmanian bolster heath string fen. Vegetatio **58**:71.

Kirkpatrick JB, Green K, Bridle KL, Venn SE. 2014. Patterns of variation in Australian alpine soils and their relationships to parent material, vegetation formation, climate and topography. Catena **121**:186–194.

Kirkpatrick JB, Marsden-Smedley JB, Di Folco MB, Leonard SWJ. 2016. Influence of grazing and vegetation type on post-fire floristic and lifeform composition in Tasmania, Australia. Plant Ecology **217**:57–69. Springer Netherlands.

Kirkpatrick JB, Nunez M, Bridle KL, Parry J, Gibson N. 2017. Causes and consequences of variation in snow incidence on the high mountains of Tasmania, 1983-2013. Australian Journal of Botany **65**:214–224.

Kitchener A, Harris S. 2013. From Forest to Fjaeldmark: Descriptions of Tasmania's VegetationEdition 2. Department of Primary Industries, Parks, Water and Environment, Tasmania.

Kivinen S, Kaarlejärvi E, Jylhä K, Räisänen J. 2012. Spatiotemporal distribution of threatened high-latitude snowbed and snow patch habitats in warming climate. Environmental Research Letters **7**.

Klose K, Cooper SD, Bennett. DM. 2015. Effects of wildfire on stream algal abundance, community structure, and nutrient limitation. Freshwater Science **34**:1494–1509.

Knott B, Suter PJ, Richardson AMM. 1978. A preliminary observation on the littoral rock fauna of Hartz Lake and Hartz Creek, Southern Tasmania, with notes on the water chemistry of some neighbouring lakes. Australian Journal of Marine and Freshwater Research **29**:703–715.

Kotze DC. 2013. The effects of fire on wetland structure and functioning. African Journal of Aquatic Science **38**:237–247.

Lamont BB, Keith DA. 2014. Heathlands and associated shrublands. Pages 339–367 in D. A. Keith, editor. Australian Vegetation, 3rd edition. Cambridge University Press, Cambridge, United Kingdom.

Langdon JS. 1989. Experimental transmission and pathogenicity of epizootic haematopoietic necrosis virus (EHNV) in redfin perch, Perca fluviatilis L., and 11 other teleosts. Journal of Fish Diseases, **12**:295–310.

Lawler SH, Crandall KA. 1998. The relationship of the Australian Freshwater crayfish genera Euastacus and Astacopsis. Proceedings of the Linnaean Society of New South Wales **119**:1–8.

Lawrence R. 1995. The Effects of Grazing Activity on the Hydrology of the Bogong High Plains, Australia. The Rangeland Journal **17**:138.

Lawrence R, Rutherfurd I, Ghadirian P, White M, Coates F, Thomas I. 2009. The geography and hydrology of high country peatlands in Victoria Part 1. Heidelberg, Victoria.

Lawrence RE. 1999. Vegetation changes on the Bogong High Plains from the 1850s to 1950s. Proceedings of the Royal Society of Victoria **111**:xxix–lii.

Lawrence RE. 2001. The impacts of hydro-electric construction works on the hydrology of a subalpine area in Australia. Environmental Geology **40**:612–621. Springer-Verlag.

Lee CKF, Keith DA, Nicholson E, Murray NJ. 2019. Redlistr: tools for the IUCN Red Lists of ecosystems and threatened species in R. Ecography **42**:1050–1055. Blackwell Publishing Ltd.

Leigh J, Wimbush D, Wood D, Holgate M, Slee A, Stanger M, Forrester R. 1987. Effects of rabbit grazing and fire on a sub-alpine environment. I. Herbaceous and shrubby vegetation. Australian Journal of Botany **35**:433–464.

Lembrechts JJ, Pauchard A, Lenoir J, Nuñez MA, Geron C, Ven A, Bravo-Monasterio P, Teneb E, Nijs I, Milbau A. 2016. Disturbance is the key to plant invasions in cold environments. Proceedings of the National Academy of Sciences of the United States of America **113**:14061–14066. Available from https://doi.org/10.1073/pnas.1608980113.

Leonard B V., Timms B V. 1974. The littoral rock fauna of three highland lakes in Tasmania. Papers and Proceedings of the Royal Society of Tasmania **108**:151–156.

Lim FKS, Pollock LJ, Vesk. PA. 2017. The Role of Plant Functional Traits in Shrub Distribution around Alpine Frost Hollows. Journal of Vegetation Science **28**:585–94. Available from https://doi.org/10.1111/jvs.12517.

Lindenmayer D, Burns E, Thurgate N, Lowe A, CSIRO (Australia), editors. 2014. Biodiversity and environmental change: monitoring, challenges, and direction. {CSIRO} Publishing.

Ling HU, Croome RL, Tyler PA. 1989. Freshwater dinoflagellates of Tasmania, a survey of taxonomy and distribution. British Phycological Journal **24**:111–129.

Lough JM, Hobday AJ. 2011. Observed Climate Change in Australian Marine and Freshwater Environments. Marine and Freshwater Research **62**:984–999. Available from https://doi.org/10.1071/MF10272.

Love J, Thapa R, Drielsma M, Robb J. 2019. Climate Change Impacts in the NSW and ACT Alpine region: Impacts on Biodiversity. Available from www.environment.nsw.gov.au.

Love PT, Fox-Hughes P, Harris R, Remenyi T, Bindoff N. 2016. Impact of Climate Change on Weather-Related Fire Risk Factors in the TWWHA: Interim Report. Hobart, Tasmania, Australia.

Lownds RM, Turbill C, White TE, Umbers KD l. 2023. The Impact of Elevated Aestivation Temperatures on the Behaviour of Bogong Moths (Agrotis Infusa). Journal of Thermal Biology **113**:103538. Available from https://doi.org/10.1016/j.jtherbio.2023.103538.

Lu Z et al. 2025. Increased frequency of multi-year El Niño–Southern Oscillation events across the Holocene. Nature Geoscience **18**:337–343.

Lucas C, Hennessy K, Mills G, Bathols J. 2007. Bushfire Weather in Southeast Australia: Recent Trends and Projected Climate Change Impacts.

Luthin JN, Guymon GL. 1974. Soil moisture - vegetation-temperature relationships in central Alaska. Journal of Hydrology **23**:233–246.

Lynch A, Kirkpatrick J. 1995. Pattern and Process in Alpine Vegetation and Landforms at Hill One, Southern Range, Tasmania. Australian Journal of Botany **43**:537–554.

Mackey B, Jacobs P, Hugh S. 2015. Classifying and mapping the Australian Alps' Native Vegetation. Cunninghamia **15**:185–199.

MacLennan MM. 2015. Cumulative Impacts of Multiple Stressors on Aquatic Communities: Novel Fish Predators in Warming Mountain Lakes. Science of the Total Environment **832**:154959.

MacPhail M. 1979. Vegetation and climates in southern Tasmania since the last glaciation. Quaternary Research **11**:306–341.

MacPhail M. 1981. A history of fire in the forest. Page in J. B. Kirkpatrick, editor. Fire and forest management in Tasmania. Tasmanian Conservation Trust, Hobart.

MacPhail M. 1986. 'Over the top' – pollen-based reconstruction of past alpine floras and vegetation in Tasmania. Pages 173–204 in B. A. Barlow, editor. Flora and Fauna of Alpine Australasia. CSIRO, Melbourne.

Maiden JH. 1898. A contribution towards a flora of Mount Kosciusko. Agricultural Gazette of N.S.W **9**:720–740.

Mallick SA, Driessen MM. 2005. An inventory of the invertebrates of the Tasmanian Wilderness World Heritage Area. Pacific Conservation Biology **11**:198–211. Available from www.dpiwe.tas.gov.au.

Marchant R, A.Graesser, Metzeling L, Mitchell P, Norris P, Suter R. 1984. Life histories of some benthic insects from the La Trobe River, Victoria. Australian Journal of Marine and Freshwater Research **35**:793–806.

Marsden-Smedley JB. 1998. Changes in southwestern Tasmanian fire regimes since the early 1800s. Papers and Proceedings of the Royal Society of Tasmania **132**:15–29.

Mason TJ, Krogh M, Popovic GC, Glamore W, Keith DA. 2021. Persistent effects of underground longwall coal mining on freshwater wetland hydrology. Science of the Total Environment **772**:144772.

McCartney VA, Silvester E, Morgan JW, Suter PJ. 2013. Physical and chemical drivers of vegetation in groundwater-source pools on the Bogong High Plains, Victoria. Australian Journal of Botany **61**:566–573.

Mccoll-Gausden SC, Bennett LT, Clarke HG, Ababei DA, Penman TD. 2022. The fuel–-climate–fire conundrum: How will fire regimes change in temperate eucalypt forests under climate change?1–16.

McColl-Gausden SC, Bennett LT, Duff TJ, Cawson JG, Penman TD. 2020. Climatic and edaphic gradients predict variation in wildland fuel hazard in south-eastern Australia. Ecography **43**:443–455.

McDougall K, Wright G, Peach E. 2018. Coming to terms with Ox-eye Daisy (Leucanthemum vulgare) in Kosciuszko National Park, New South Wales. Ecological Management & Restoration **19**:4–13.

McDougall KL. 1982. The alpine vegetation of the Bogong High Plains. Soil Conservation Authority, Victoria, Kew.

McDougall KL. 1989. The effect of excluding cattle from a mossbed on the Bogong High Plains, Victoria.

McDougall KL. 2003. Aerial photographic interpretation of vegetation changes on the Bogong High Plains, Victoria, between 1936 and 1980. Australian Journal of Botany **51**:251–256.

McDougall KL. 2007. Grazing and fire in two subalpine peatlands. Australian Journal of Botany **55**:42–47.

McDougall KL, Broome LS. 2007. Challenges facing protected area planning in the Australian Alps in a changing climate. In Protected Areas: Buffering nature against climate change. Pages 73–84 Proceedings of a WWF-Australia and IUCN World Commission on Protected Areas symposium,. Canberra.

McDougall KL, Morgan JW, Walsh NG, Williams RJ. 2005. Plant invasions in treeless vegetation of the Australian Alps. Perspectives in Plant Ecology, Evolution and Systematics **7**:159–171.

McDougall KL, Walsh NG. 2007. Treeless vegetation of the Australian Alps. Cunninghamia **10**:1–57. Available from https://www.researchgate.net/publication/267835588.

McDougall KL, Walsh NG, Wright GT. 2015. Recovery of treeless subalpine vegetation in Kosciuszko National Park after the landscape-scale fire of 2003. Australian Journal of Botany **63**:597–607.

McDougall KL, Whinam J, Coates F, Morgan JW, Walsh NG, Wright GT, Hope GS. 2023. Fire in the bog: responses of peatland vegetation in the Australian Alps to fire. Australian Journal of Botany **71**:111–126.

McDougall KL, Wright GT. 2004. The impact of trampling on fjaeldmark vegetation in Kosciuszko National Park, New South Wales. Australian Journal of Botany **52**:315–320.

McIntosh AR, McHugh PA, Dunn NR, Goodman JM, Howard SW, Jellyman PG, O'Brien LK, Nyström P, Woodford DJ. 2010. The impact of trout on galaxiid fishes in New Zealand. New Zealand Journal of

Ecology **34**:195 – 206.

McMahon A, Carr G, Sutton F. 2009. Fainter Ridge Willow Infestation. Report to Parks Victoria, Ecology Australia.

McVean DN. 1969. Alpine vegetation of the central Snowy Mountains of New South Wales. Journal of Ecology:67–86.

Meaher D, B. Fuhrer. 2003. A Field Guide to the Mosses and allied Planrs of Southern Australia. Australian Biological Resources Study and Field Naturalists Club of Victoria, Canberra.

Meier W, Bonjour C, Wüest A, Reichert P. 2003. Modeling the Effect of Water Diversion on the Temperature of Mountain Streams. Journal of Environmental Engineering **129**. Available from https://doi.org/10.1061/(ASCE)0733-9372(2003)129:8(755).

Minchin PR, Davies JB. 2019. Effects of climate change on the alpine and subalpine vegetation of the Mt. Field National Park, Australia. Page British Ecological Society Annual Meeting, Belfast, Northern Ireland, December 10-13, 2019.

Minchin PR, Davies JB, Oksanen J. 2022. Upward altitudinal shifts of plant species at Mt. Field, Australia in response to a warming climate. Page British Ecological Society Annual Meeting, Edinburgh, Scotland, December 18-21, 2022.

Monaghan MT, Robinson CT, Spaak P, Ward J V. 2005. Macroinvertebrate diversity in fragmented Alpine streams: implications for freshwater conservation. Aquatic Sciences **67**:454–464.

Morgan J, Green P. 2013. Vegetation Change in the Pretty Valley Alpine Grasslands from 1945 to 2003: The Influence of Grazing on Plant Community Succession. Bundora, Victoria.

Morgan J, Walker Z. 2023. Early-melting snowpatch plant communities are transitioning into novel states. Scientific Reports **13**:16520.

Morgan JW, Shackleton M, Walker ZC. 2024. Long-unburnt stands of snow gum (Eucalyptus pauciflora Sieber ex Spreng) are exceedingly rare in the Victorian Alps: Implications for their conservation and management. Australian Journal of Botany **72**. Available from https://doi.org/10.1071/BT23068.

Morgan JW, Venn SE. 2017. Alpine plant species have limited capacity for long-distance seed dispersal. Plant Ecology **218**:813–819. Springer Netherlands.

Mosley LM. 2015. Drought impacts on the water quality of freshwater systems; review and integration. Earth-Science Reviews **140**:203–214.

Moss S. 2024. Multi-Decadal Changes in Alpine Grasslands: The Interaction Between Land-Use Legacy and Climate Change. La Trobe University. Available from https://opal.latrobe.edu.au/articles/thesis/Multi-decadal_Changes_in_Alpine_Grasslands_The_Interaction_Between_Land-use_Legacy_and_Climate_Change/27140109?file=49520778.

MountainWatch. 2023a. Ben Lomond. Ben Lomond Ski Resort | Ski Resorts Australia | Mountainwatch.

MountainWatch. 2023b. Mount Mawson.

Murphy BP, Prior LD, Cochrane MA, Williamson GJ, Bowman DMJS. 2019. Biomass consumption by surface fires across Earth's most fire prone continent. Global Change Biology **25**:254–268. Available from https://doi.org/10.1111/gcb.14460.

Myers-Smith IH et al. 2011. Shrub expansion in tundra ecosystems: Dynamics, impacts and research priorities. Environmental Research Letters **6**.

Naccarella A, Morgan JW, Cutler SC, Venn SE. 2020. Alpine treeline ecotone stasis in the face of recent climate change and disturbance by fire. PLoS ONE **15**:e0231339. Available from https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0231339.

National Waterbug Blitz Team. 2025. Waterbug Data Portal. Federation University, CERDI. Available from https://www.waterbugblitz.org.au/explore.php.

Natural Values Conservation Branch. 2017. Assessment of the ecological impacts of the 2016 Mersey Forest Fire Complex, Nature Conservation Report no 17/5. Hobart.

New TR. 2011. Launching and steering flagship Lepidoptera for conservation benefit. Journal of Threatened Taxa **3**:1805–1817.

Newell G, White M, Griffioen P, Conroy M. 2006. Vegetation condition mapping at a landscape-scale across Victoria. Ecological Management & Restoration **7**:S65–S68.

NHT. 2003. Weeds of National Significance: Weed Management Guide, Willow - Salix spp.

Nichols EC, Kirkpatrick JB. 2019. Patterns and causes of marsupial paths in subalpine Tasmania. Papers and Proceedings of the Royal Society of Tasmania **153**:53–60. Royal Society of Tasmania.

Nielsen E, McQuillan P, Common I. 1992. The Tasmanian cushion plant moth Nemotyla oribates: systematics and biology (Lepidoptera: Oecophoridae: Xyloryctinae). Journal of the Australian

Entomological Society **31**:47–56.

Nimmo DG, Miller KK. 2007. Ecological and human dimensions of management of feral horses in Australia: A review. Wildlife Research **34**:408–417.

Nolan RH et al. 2021. Hydraulic failure and tree size linked with canopy die-back in eucalypt forest during extreme drought. New Phytologist **230**:1354-1365. Available from https://doi.org/10.1111/nph.17298.

Nolan RH, Boer MM, Dios VR de, Caccamo G, Bradstock RA. 2016. Large-scale, dynamic transformations in fuel moisture drive wildfire activity across southeastern Australia. Geophysical Research Letters **43**:4229–4238. Available from https://doi.org/10.1002/2016GL068614.

Normyle A, Pittock J. 2020. A review of the impacts of pumped hydro energy storage construction on subalpine and alpine biodiversity: lessons for the Snowy Mountains pumped hydro expansion project. Australian Geographer **51**:53–68. Taylor & Francis. Available from https://doi.org/10.1080/00049182.2019.1684625.

NPWS. 2007. Kosciuszko National Park Horse Management Plan Draft.

NSW Goverment. 2022. NPWS Fire History - Wildfires and Prescribed Burns Dataset. State Government of NSW and Department of Planning and Environment. Available from https://datasets.seed.nsw.gov.au/dataset/fire-history-wildfires-and-prescribed-burns-1e8b6.

NSW OEH. 2019. Windswept Fjaeldmark in the Australian Alps Bioregion – profile. Department of Planning, Industry and Environment, Office of Environment and Heritage. Available from https://www.environment.nsw.gov.au/threatenedspeciesapp/profile.aspx?id=20297.

NSW Threatened Species Scientific Committee. 2018. Windswept Feldmark in the Australian Alps Bioregion - critically endangered ecological community listing **1995**:23–26. Available from https://www.environment.nsw.gov.au/topics/animals-and-plants/threatened-species/nsw-threatened-species-scientific-committee/determinations/final-determinations/2017-2018/snowpatch-herbfield-in-the-australian-alps-critically-endangered-ecological-community.

Nunez M, Kirkpatrick JB, Nilsson C. 1996. Rainfall estimation in south-west Tasmania using satellite images and phytosociological calibration. International Journal of Remote Sensing **17**:1583–1600.

NVIS Technical Working Group. 2017. Australian Vegetation Attribute Manual: National Vegetation Information System, Version 7.0. Page (Bolton MP, DeLacey C, Bossard KB, editors). Department of the Environment and Energy, Canberra.

OEH. 2011. New South Wales Climate Impact Profile Technical Report: Potential impacts of climate change on biodiversity.

Otley H, Munks S, Hindel M. 2000. Activity patterns, movements and burrows of platypuses (Ornithorhynchus anatinus) in a sub-alpine Tasmanian lake. Australian Journal of Zoology **48**:701–713.

Pancotto V, Holl D, Escobar J, Castagnani M, Kutzbach L. 2021. Cushion bog plant community responses to passive warming in southern Patagonia. Biogeosciences **18**:4817–4839.

Paoletti M, Osler G, Kinnear A, Black D, Thomson L, Tsitsilas A, Sharley D, Judd S, Neville P, D'Inca A. 2007. Detritivores as indicators of landscape stress and soil degradation. Australian Journal of Experimental Agriculture **47**:412–23.

Parry J, Balmer J. 2017. Establishment Report for the Tasmanian Wilderness World Heritage Area Climate Change Monitoring Program Snow patch vegetation on Cradle Mountain and Mount Eliza, Tasmania. DPIPWE, Tasmanian Government.

Parry J, Kirkpatrick JB, Marsden-Smedley J. 2016. Explaining the distribution, structure and species composition of snow-patch vegetation in Tasmania, Australia. Australian Journal of Botany **64**:484–491.

PBPL. 2002. Perisher Blue Ski Resort Ski Slope Master Plan. Perisher Valley: Perisher Blue Pty Ltd.

Pebesma E, Bivand R. 2023. Spatial Data Science: With applications in R. Chapman and Hall/CRC. Available from https://r-spatial.org/book/.

Penman TD, Ababei D, Chong DMO, Duff TJ, Tolhurst KG. 2015. A fire regime risk management tool. Pages 270– 276 in J. Sharples, editor. 21st international congress on modelling and simulation, Gold Coast, Australia, 29 Nov to 4 Dec 2015. Available from www.mssanz.org.au/modsim2015.

Pepler AS, Trewin B, Ganter C. 2015. The influences of climate drivers on the Australian snow season. Australian Meteorological and Oceanographic Journal **65**:2–11.

Perga ME, Bruel R, Rodriguez L, Guénand Y, Bouffard D. 2018. Storm impacts on alpine lakes: Antecedent weather conditions matter more than the event intensity. Global Change Biology, **24**:5004–5016.

Petitpierre B, McDougall K, Seipel T, Broennimann O, Guisan A, Kueffer C. 2016. Will climate change increase the risk of plant invasions into mountains? Ecological Applications **26**:530–544.

Pickering C. 2007. Climate change and other threats in the Australian Alps. Page Proceedings of the WWF and IUCN World Commission on Protected Areas Symposium. Available from http://hdl.handle.net/10072/19118http://wwf.org.au/.

Pickering C, Armstrong T. 2003. The potential impacts of climate change on plant communities in the Kosciuszko alpine zone. The Victorian Naturalist **120**.

Pickering C, Armstrong T. 2014. Climate change and the plant communities of the Kosciuszko alpine zone in the Australian Alps. Cooperative Research Centre for Sustainable Tourism: Griffith University.

Pickering C, Good R, Green K, Australian Greenhouse Office. 2004. Potential Effects of Global Warming on the Biota of the Australian Alps. Australian Greenhouse Office, Canberra.

Pickering C, Green K, Barros AA, Venn S. 2014. A resurvey of late-lying snowpatches reveals changes in both species and functional composition across snowmelt zones. Alpine Botany **124**:93–103.

Pickering C, Michael A. 2022. Nature-based tourists as seed dispersal vectors. Tourism, Recreation and Biological Invasions:7–13. Available from https://doi.org/10.1079/9781800620544.0002.

Pickering CM, Barry K. 2005. Size/age distribution and vegetative recovery of Eucalyptus niphophila (snowgum, Myrtaceae) one year after fire in Kosciuszko National Park. Australian Journal of Botany **53**:517–527.

Pickering CM, Venn S. 2013. Increasing the Resilience of Australian Alpine Flora to Climate Change and Assosiated Threats: A Plant Functional Traits Approach.

Ponder W, Chilcott S, Fulton W, Kirkpatrick J, Richardson A. 1993. Streams and Rivers. Pages 114–122 in S. Smith and M. Banks, editors. 1993: Tasmanian Wilderness – World Heritage Values. Royal Society of Tasmania, Hobart.

Preston DL, Caine N, McKnight DM, Williams MW, Hell K, Miller MP, Hart SJ, Johnson PTJ. 2016. Climate regulates alpine lake ice cover phenology and aquatic ecosystem structure. Geophysical Research Letters **43**.

Prior LD, French BJ, Storey K, Williamson GJ, Bowman DMJS. 2020. Soil moisture thresholds for combustion of organic soils in western Tasmania. International Journal of Wildland Fire **29**:637–647. CSIRO.

Prior LD, Nichols SC, Williamson GJ, Bowman DMJS. 2022. Post-fire restoration of Sphagnum bogs in the Tasmanian Wilderness World Heritage Area, Australia . Restoration Ecology:1–13.

Prior LD, Storey K, Williamson GJ, Bowman DM. 2024. When soil becomes fuel: identifying a safe window for prescribed burning of Tasmanian vegetation growing on organic soils. International Journal of Wildland Fire **33**:WF24061.

Prober SM, Thiele KR. 2007. Assessment of Impacts of Feral Horses (Equus caballus) in the Australian Alps.

Prosser IP, Williams L. 1998. The effect of wildfire on runoff and erosion in native Eucalyptus forest. Hydrological Processes **12**:251–265.

R Core Team. 2023. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. Available from https://www.r-project.org/.

Raadik TA. 2014. Fifteen from one: a revision of the Galaxias olidus Günther, 1866 complex (Teleostei, Galaxiidae) in south-eastern Australia recognises three previously described taxa and describes 12 new species. Zootaxa **3898**:1–198.

Read J, Hill R. 1988. Comparative responses to temperature of the major canopy species of Tasmanian cool temperate rainforest and their ecological significance. I. Foliar Frost-Resistance. Australian Journal of Botany **35**:131–143.

Regan TJ, Tolsma A, Rowland J, Muir A, Ferrer-Paris JR, Tóth AB, White M. 2020. Risk assessment and management priorities for alpine ecosystems under climate change: Milestone 5 Report. Heidelberg, Victoria.

Reid JB, Potts BM. 1999. Eucalypt Biology. Vegetation of Tasmania:198–223.

Reisinger A, Kitching RL, Chiew F, Hughes L, Newton PCD, Schuster SS, Tait A, P.Whetton. 2014. Australasia. Pages 1371–1438 in V. R. Barros et al., editors. Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Reneau SL, Katzman D, Kuyumjian GA, Lavine A, Malmon DV. 2007. Sediment delivery after a wildfire. Geology **35**:151–154.

Richards RM, Cary GJ, Bradstock RA. 2001. The sensitivity of snow gum to fire scarring in relation to

Aboriginal landscape burning. Pages 285–92 Australasian Bushfire Conference. Christchurch.

Richardson D, Black AS, Monselesan DP, Risbey JS, Squire DT, Tozer CR, Canadell JG. 2021. Increased extreme fire weather occurrence in southeast Australia and related atmospheric drivers. Weather and Climate Extremes **34**:100397. Elsevier B.V. Available from https://doi.org/10.1016/j.wace.2021.100397.

Rigg JL, McDougall KL, Liew ECY. 2018. Susceptibility of nine alpine species to the root rot pathogens Phytophthora cinnamomi and P. cambivora. Australasian Plant Pathology **47**:351–356. Australasian Plant Pathology.

Rivrud IM, Meisingset EL, Loe LE, Mysterud A. 2019. Future suitability of habitat in a migratory ungulate under climate change. Proceedings. Biological sciences **286**:20190442.

Roberts C, Kirkpatrick J, McQuillan P. 2011. Tasmanian lentic wetland lawns are maintained by grazing rather than inundation. Austral Ecology **36**:303–309.

Roberts I. 1996. Baw Baw Plateau Vegetation Map. Earth Resource Analysis, Forrest, Victoria. Available from https://www.vgls.vic.gov.au/client/en_AU/vgls/search/detailnonmodal/ent:$002f$002fSD_ILS$002f0$002fSD_ILS:23617/ada?qu=Victoria.+Ministry+for+Conservation.&d=ent%3A%2F%2FSD_ILS%2F0%2FSD_ILS%3A23617~ILS~207&ic=true&ps=300&h=8.

Robertson G, Wright J, Brown D, Yuen K, Tongway D. 2019. An assessment of feral horse impacts on treeless drainage lines in the Australian Alps. Ecological Management and Restoration **20**:21–30. Blackwell Publishing.

Rotvit L, Jacobsen D. 2014. Egg development of Plecoptera, Ephemeroptera and Odonata along latitudinal gradients. Ecological Entomology **39**:177–185.

Rydin H, Jeglum JK. 2013. The Biology of Peatlands, 2nd Edition. Oxford University Press, Oxford.

Rydin H, Jeglum JK. 2015. The Biology of Peatlands. Page The Biology of Peatlands. Oxford University Press.

Saber A, James DE, Hannoun IA. 2020. Effects of lake water level fluctuation due to drought and extreme winter precipitation on mixing and water quality of an alpine lake, Case Study: Lake Arrowhead, California. Science of the Total Environment **714,**:136762.

Saft M, Peel M, Peterson T. 2020. Characterising transition towards more ephemeral streams in Australian catchments. EGU General Assembly 2020:EGU2020-21214. Available from https://doi.org/10.5194/egusphere-egu2020-21214.

Salmona J, Dixon KM, Banks SC. 2018. The effects of fire history on hollow-bearing tree abundance in montane and subalpine eucalypt forests in southeastern Australia. Forest Ecology and Management, **428**:93–103.

Sánchez-Bayo F, Green K. 2013. Australian snowpack disappearing under the influence of global warming and solar activity. Arctic, Antarctic, and Alpine Research **45**:107–118.

Sato CF, Schroder M, Green K, Michael DR, Osborne WS, Lindenmayer DB. 2014. Managing ski resorts to improve biodiversity conservation: Australian reptiles as a case study. Ecological Management & Restoration **15**:147–154. Available from https://doi.org/10.1111/emr.12111.

Scherrer P, Pickering CM. 2005. Recovery of alpine vegetation from grazing and drought: Data from long-term photoquadrats in Kosciuszko National Park, Australia. Arctic, Antarctic, and Alpine Research **37**:574–584.

Scherrer P, Wimbush D, Wright G. 2004. The assessment of pre and post 2003 wildfire data collected from subalpine transects in Kosciuszko National Park. Report to Department of Environment and Conservation. Sydney.

Schroder M. 2014. Processes Explaining Exotic Plant Occurence in Australian Mountain Systems. Charles Sturt University. Available from https://researchoutput.csu.edu.au/ws/portalfiles/portal/9312924/63880.

Schroder M, Burgess H, Greville T, McDougall KL. 2015. A collaborative approach to the impacts of a changing climate: Early detection of emerging weeds in the Australian Alps. Plant Protection Quarterly **30**:93.

Schulz M, Schroder M, Green K. 2019. The occurrence of the Broad-toothed Rat Mastacomys fuscus in relation to feral Horse impacts. Ecological Management and Restoration **20**:31–36. Blackwell Publishing.

Scott D, McBoyle G. 2007. Climate change adaptation in the ski industry. Mitigation and Adaptation Strategies for Global Change **12**:1411–1431. Available from https://link.springer.com/article/10.1007/s11027-006-9071-4.

Seaton S, Matusick G, Ruthrof KX, Hardy GESJ. 2015. Outbreak of Phoracantha semipunctata in response to severe drought in a Mediterranean Eucalyptus forest. Forests **6**:3868–3881.

SEWPaC. 2012. Interim Biogeographic Regionalisation for Australia, Version 7. Department of Sustainability, Environment, Water, Population and Communities. Available from http://www.environment.gov.au/parks/nrs/science/bioregion-framework/ibra/maps.html.

Shakesby RA, Doerr SH. 2006. Wildfire as a hydrological and geomorphological agent. Earth-Science Reviews **74**:269–307.

Shields B. 1993. Patch Dieback of the Subalpine Snow Gum Forest, E. pauciflora in Kosciusko National Park. Australian National University.

Short LE, Gabet EJ, Hoffman DF. 2015. The role of large woody debris in modulating the dispersal of a post-fire sediment pulse. Geomorphology **246**:351–358. Elsevier B.V. Available from http://dx.doi.org/10.1016/j.geomorph.2015.06.031.

Siebers AR, Paillex A, Misteli B, Robinson CT. 2020. Effects of an experimental increase in flow intermittency on an alpine stream. Hydrobiologia **847**:3453–3470.

Silvester E. 2006. Monitoring Water Quality and Biogeochemical Changes During Peatland Rehabilitation. Albury-Wodonga.

Silvester E. 2007. Diurnal Patterns in Stream Water Constituents as Indicators of Peatland Condition. Albury-Wodonga.

Silvester E. 2009. Ionic regulation in an alpine peatland in the Bogong High Plains, Victoria, Australia. Environmental Chemistry **6**:424–431.

Slatyer R. 2010. Climate change impacts on Australia’s alpine ecosystems. ANU Undergraduate Research Journal **2**:81–97.

Slee A, Shulmeister J, Kiernan K, Jenkinson A. 2016. Stone-banked lobes as a product of mild freeze–thaw action: an example from Western Tasmania, Australia. Geografiska Annaler: Series A, Physical Geography **98**:97–109.

Snowy Hydro Limited. 2020. History. Available from https://www.snowyhydro.com.au/about/history/ (accessed July 18, 2023).

SOSnowgum. 2023. SOSnowgum: Report Dieback. Available from https://www.saveoursnowgum.org/help-us.

Specht RL. 1970. Vegetation. Pages 44-67. in G. W. Leeper, editor. Australian environment4th Ed. CSIRO-Melbourne University Press, Melbourne.

Specht RL. 1981. Foliage projective cover and standing biomass. Pages 10–21. in A. N. Gillison and D. J. Anderson, editors. Vegetation Classification in Australia. Canberra.

Steane D, Tolsma A, Papst W. 2005. A survey of the Distribution and Habitat Preference of the Alpine Water Skink Eulamprus kosciuskoi on the Bogong High Plains, Victoria. Report to Parks Victoria.

Stern H, De Hoedt G, Ernst J. 2000. Objective classification of Australian climates. Australian Meteorological Magazine **49**:87–96.

Stoessel DJ, Raadik TA, Ayres RM. 2015. Spawning of threatened barred galaxias,’Galaxias fuscus’(Teleostei: Galaxiidae). Proceedings of the Linnean Society of New South Wales **137**:1–6.

Strahler A. 1952. Hypsometric (area–altitude) analysis of erosional topography. Geological Society of America Bulletin **63**:1117–1141.

Strahler AN. 1957. Quantitative analysis of watershed geomorphology. Eos, Transactions American Geophysical Union **38**:913–920.

Sturm M, McFadden JP, Liston GE, Stuart Chapin F, Racine CH, Holmgren J. 2001. Snow-shrub interactions in Arctic Tundra: A hypothesis with climatic implications. Journal of Climate **14**:336–344.

Styger J, Marsden-Smedley J, Kirkpatrick J. 2018. Changes in lightning fire incidence in the tasmanian wilderness world heritage area, 1980–2016. Fire **1**:1–10. MDPI AG.

Sumner EE, Venn SE. 2022. Thermal tolerance and growth responses to in situ soil water reductions among alpine plants. Plant Ecology and Diversity **15**:297–308.

Suter P, Webb J. 2012. Key to the mature nymphs of Ameletoides (Ephemeroptera : Nesameletidae). Pages 1–11 in P. Suter, editor. Taxonomy Research and Information Network Taxonomic Workshop 2012. La Trobe University, Wodonga Victoria.

Suter PJ, Bishop JE. 1990. Stoneflies (Plecoptera) of South Australia. Series Entomologica **44**:189–207.

Suter PJ, Clair R St., Hawking J, Bryce C. 2002. Aquatic macroinvertebrates from streams in the Mt. Kosciuszko area. Pages 90–97 in K. Green, editor. Biodiversity in the Snowy Mountains. National Parks and Wildlife Service, Australian Institute of Alpine Studies, Jindabyne, Australia.

Suter PJ, McGuffie PJ. 2007. Conservation of mayflies (Ephemeroptera) especially Coloburiscoides in the Victorian Alps: impediments and threats. Victorian Naturalist **124**:273–277.

Suter SG, Rees GN, Watson GO, Suter PJ, Silvester E. 2011. Decomposition of native leaf litter by aquatic hyphomycetes in an alpine stream of south-eastern Australia. Marine and Freshwater Research **62**:841–849.

Suter SGA. 2014. Contribution of aquatic Hyphomycete communities to leaf decomposition and food webs in alpine streams. La Trobe University.

Sweeney BW, Vannote RL. 1978. Size variation and distribution of hemimetabolous aquatic insects: two thermal equilibrium hypotheses. Science **200**:444–446.

Tasmanian Goverment. 2014. TASVEG 3.0. Available from https://www.thelist.tas.gov.au/app/content/data/geo-meta-data-record?detailRecordUID=6e3d672d-cdc1-40f3-8ee4-b45c5de68b68.

Tasmanian Government. 2022. Fire History. Available from https://listdata.thelist.tas.gov.au/opendata/.

Tasmanian Government. 2024. LIST Fire History. Available from https://listdata.thelist.tas.gov.au/opendata/.

Thomas I, Hope G. 1994. An example of Holocene vegetation stability from Camerons Lagoon, a near treeline site on teh Central Plateau, Tasmania. Australian Journal of Ecology **19**:150–158.

Thomas V, Gellie N, Harrison T. 2000. Forest Ecosystem Classification and Mapping for the Southern CRA Region. A Report Undertaken for the NSW Comprehensive Regional Assessment/Regional Forest Agreement Steering Committee. Queanbeyan.

Threatened Species Scientific Committee. 2009. Commonwealth Listing Advice on Alpine Sphagnum Bogs and Associated Fens. Available from http://www.environment.gov.au/biodiversity/threatened/communities/pubs/29-listing-advice.pdf.

Threatened Species Scientific Committee. 2011. Approved Conservation Advice for Thaumatoperla alpina (Alpine Stonefly). Available from http://www.environment.gov.au/biodiversity/threatened/species/pubs/25289-conservation-advice.pdf.

Tibby J, Tyler J, Barr C. 2018. Post little ice age drying of eastern Australia conflates understanding of early settlement impacts. Quaternary Science Reviews **202**:45–52. Available from doi.org/10.1016/j.quascirev.2018.10.033.

Tilzey R. 1980. Introduced fish. Pages 271–279 in W. Williams, editor. An Ecological Basis for Water Resource Management. Australian National University Press, Canberra.

Timms B V. 1980. The benthos of Australian lakes. Pages 23–39 in W. D. Williams, editor. An Ecological Basis for Water Resource Management. ANU Press, Canberra.

Timms B V. 2002. Lake Cootapatamba. Pages 98–101 in K. Green, editor. Biodiversity in the Snowy Mountains. National Parks and Wildlife Service, Australian Institute of Alpine Studies, Jindabyne, Australia.

Timms B V., Morton J, K. Green. 2013. Temporal changes in the Macroinvertebrate Fauna of two Glacial Lakes, Cootapatamba and Albina, Snowy Mountains, New South Wales. Proceedings of the Linnean Society of New South Wales **135**:45–54.

Tolhurst K, Shields B, Chong D. 2008. Phoenix: Development and application of a bushfire risk management tool. The Australian Journal of Emergency Management **23**:47– 54.

Tolsma A. 2008a. An Assessment of the Condition of Mossbeds in State Forest, Gippsland, Victoria. Report to the Department of Sustainability and Environment. Heidelberg.

Tolsma A. 2008b. An Assessment of the Management Needs of Mossbeds in Victoria's Alps, 2004-2008. Report to Parks Victoria. Heidelberg.

Tolsma A. 2009. An assessment of mossbeds across the Victorian Alps, 2004–2009.

Tolsma A. 2020. Bushfire Biodiversity Response and Recovery Program: Post-fire assessment of Alpine Bogs. Published Client report for DELWP. Heidelberg, Victoria.

Tolsma A, Shannon J. 2007. Evaluating the Rehabilitation Needs of Mossbeds in the Alpine and Mt Buffalo National Parks After the 2006/07 Fires. Unpublished report to Parks Victoria.

Tolsma A, Shannon J. 2018. Assessing the Impacts of Feral Horses on the Bogong High Plains, Victoria. Unpublished client report for Parks Victoria, Department of Environment, Land, Water and Planning, Heidelberg, Victoria:29. Available from www.relayservice.com.au.

Tolsma A, Shannon J, Papst W, Rowe K, Rosengren N. 2005. An assessment of the condition of mossbeds on the Bogong High Plains.

Tran BN, Tanase MA, Bennett LT, Aponte C. 2020. High-severity wildfires in temperate Australian forests have increased in extent and aggregation in recent decades. PLoS ONE **15**:e0242484. Available from https://dx.plos.org/10.1371/journal.pone.0242484 (accessed March 22, 2024).

Treby S, Grover S. 2023. Carbon emissions from Australian Sphagnum peatlands increase with feral horse (Equus caballus) presence. Journal of Environmental Management **347**:119034.

Treby S, Venn S, Grover S. 2024. Exploring the Physical Properties of Australian Alpine Soils to Inform Ecosystem Restoration. Geoderma Regional **39**:e00896. Available from https://doi.org/10.1016/j.geodrs.2024.e00896.

TSSC. 2018. Index to Final Determinations 1996-2018. NSW Threatened Species Scientific Committee.

Tyler P. 1974. Limnological studies. Pages 29–61 in W. D. Williams, editor. Biogeography and Ecology in Tasmania. Monographia Biologicae.

Tyler PA. 1992. "A lakeland from the dreamtime" the second founders' lecture. British Phycological Journal **27**:353–368.

Udy DG, Vance TR, Kiem AS, Holbrook NJ, Abram N. 2004. Australia's 2019/20 Black Summer Fire Weather Exceptionally Rare over the Last 2000 Years. Communications Earth & Environment **5**:317. Available from https://doi.org/10.1038/s43247-024-01470-z.

UNEP. 1992. World Atlas of Desertification.

Van Beest C, Petrone R, Nwaishi F, Waddington JM, Macrae M. 2019. Increased peatland nutrient availability following the fort McMurray Horse River wildfire. Diversity **11**:142.

van Breemen N. 1995. How Sphagnum bogs down other plants. Tree **10**:270–5.

Van Klinken RD, Friedel MH. 2017. Unassisted invasions: Understanding and responding to Australia's high-impact environmental grass weeds. Australian Journal of Botany **65**:678–690.

Vanhoutte K, Verleyen E, Kilroy C, Sabbe K, Dasseville R, Vyvermand W. 2006. Catchment characteristics and chemical limnology of small lakes, tarns and mire pools in New Zealand (South Island) and Tasmania. Marine and Freshwater Research **57**:83–93.

Venn S, Gallagher R, Nicotra A. 2021. Germination at Extreme Temperatures: Implications for Alpine Shrub Encroachment. Plants **10**:327.

Venn S, Morgan J, Green P. 2009. Do facilitative interactions with neighboring plants assist the growth of seedlings at high altitudes in alpine Australia? Arctic, Antarctic, and Alpine Research **41**:381–387.

Venn SE, Green K. 2018. Evergreen alpine shrubs have high freezing resistance in spring, irrespective of snowmelt timing and exposure to frost: an investigation from the Snowy Mountains, Australia. Plant Ecology **219**:209–216.

Venn SE, Green K, Pickering CM, Morgan JW. 2011. Using plant functional traits to explain community composition across a strong environmental filter in Australian alpine snow patches. Plant Ecology **212**:1491–9.

Venn SE, Kirkpatrick J, McDougall KL, Walsh NG, Whinam J, Williams RJ. 2017. Alpine, Sub-alpine and Sub-Antarctic Vegetation of Australia. Pages 461–489 in D. A. Keith, editor. Australian Vegetation3rd edn. Cambridge University Press, Cambridge, United Kingdom.

Venn SE, Morgan JW. 2007. Phytomass and phenology of three alpine snowpatch species across a natural snowmelt gradient. Australian Journal of Botany **55**:450–456.

Venn SE, Morgan JW, Lord JM. 2013. Foliar freezing resistance of Australian alpine plants over the growing season. Austral Ecology **38**:152–161.

Venn SE, Pickering CM, Butler SA, Letten AD. 2016. Using a model based fourth-corner analysis to explain vegetation change following an extraordinary fire disturbance. Oecologia **182**:855–863.

Verrall B. 2018. Recovery of snow dependent plant communities in Australia from fire. Griffith University.

Verrall B. 2023. Assessing alpine vegetation dynamics using long-term ecological monitoring amidst rapid climate change. Griffith. Available from https://research-repository.griffith.edu.au/server/api/core/bitstreams/e741c759-52fc-4061-90a5-f763b5aa8677/content.

Verrall B, Green K, Pickering CM. 2023. Temporal dynamics in alpine snowpatch plants along a snowmelt gradient explained by functional traits and strategies. Oecologia **201**:155–171.

Verrall B, Pickering CM. 2019. Recovery of subalpine grasslands 15 years after landscape level fires. Australian Journal of Botany **67**:425–436.

Verrall B, Pickering CM. 2020. Alpine vegetation in the context of climate change: A global review of past research and future directions. Science of The Total Environment **748**:141344. Elsevier B.V.

Victorian Government. 2022a. Fire History Records of Fires across Victoria showing the fire scars. Department of Environment, Land, Water & Planning. Available from https://discover.data.vic.gov.au/dataset/fire-history-records-of-fires-across-victoria-showing-the-fire-scars1.

Victorian Government. 2022b. Fire history overlay of most recent fires. Department of Environment, Land, Water & Planning. Available from https://discover.data.vic.gov.au/dataset/fire-history-overlay-of-most-recent-fires.

Visoiu M, Balmer J. 2024. Montane Conifer Condition Monitoring, Revisit 1 – 2011 / 2012 to 2022 / 2023, TWWHA Natural Values Climate Change Adaptation and Resilience Program. Nature Conservation Report 2024 / 04. NRE Tas, Hobart.

Visoiu M, Whinam J. 2015. Extreme weather conditions correspond with localised vegetation death at Cradle Mountain, Tasmania. Ecological Management and Restoration **16**:76–78. Available from http://www.bom.gov.au/climate/.

Visser ME, Both C. 2005. Shifts in phenology due to global climate change: The need for a yardstick. Proceedings of the Royal Society B: Biological Sciences **272**:2561–2569. Royal Society.

Wahren C-HA. 1997. Vegetation dynamics on the Bogong High Plain. Monash University.

Wahren C-HA, Williams RJ, Papst WA. 2001. Vegetation Change and Ecological Processes in Alpine and Subalpine Sphagnum Bogs of the Bogong High Plains, Victoria, Australia. Arctic, Antarctic, and Alpine Research **33**:357–368.

Wahren C-HH, Williams RJ, Papst WA. 1999. Alpine and aubalpine wetland vegetation on the Bogong High Plains, south-eastern Australia. Australian Journal of Botany **47**:165–188.

Wahren CH, Camac JS, Jarrad FC, Williams RJ, Papst WA, Hoffmann AA. 2013. Experimental warming and long-term vegetation dynamics in an alpine heathland. Australian Journal of Botany **61**:36–51.

Wahren CHA, Papst WA, Williams RJ. 1994. Long-term vegetation change in relation to cattle grazing in sub-alpine grassland and heathland on the bogong high-plains:An analysis of vegetation records from 1945 to 1994. Australian Journal of Botany **42**:607–639.

Walsh NG, Barley RH, Gullan. PK. 1984. The alpine vegetation of Victoria (excluding the Bogong High Plains region). Dept. of Conservation, Forests and Lands.

Walsh NG, Mcdougall KL. 2004. Progress in the recovery of the flora of treeless subalpine vegetation in Kosciuszko National Park after the 2003 fires. Cunninghamia **63**:597–607. Available from https://www.researchgate.net/publication/268377854.

Ward-Jones J, Pulsford I, Thackway R, Bishwokarma D, Freudenberger D. 2019. Impacts of feral horses and deer on an endangered woodland of Kosciuszko National Park. Ecological Management and Restoration **20**:37–46. Blackwell Publishing.

Ward J V. 1994. Ecology of alpine streams. Freshwater Biology **32**:277–294.

Waterhouse B. 1988. Broom (Cytisus scoparius) at Barrington Tops, New South Wales. Australian Geographical Studies **26**:239–248.

Wearne L, Morgan J. 2004. Community level changes in Australian subalpine vegetation following invasion by the non-native shrub Cytisus scoparius. Journal of Vegetation Science **15**:595–604.

WeedFutures. 2023. WeedFutures: Determining current and future weed threats in Australia.

Western AW, Sirawardena L, Lawrence R, Rutherfurd I. 2008. Sponges or wicks? What is the role of bogs in hydrological response on the Bogong High Plains? Page Water Down Under 2008. Adelaide.

Whinam J, Barmuta L., Chilcott N. 2001. Floristic description and environmental relationships of Tasmanian Sphagnum communities and their conservation management. Australian Journal of Botany **49**:673–685.

Whinam J, Buxton R. 1997. Sphagnum peatlands of Australasia: an assessment of harvesting sustainability. Biological Conservation **82**:21–9.

Whinam J, Chilcott N. 2002. Floristic description and environmental relationships of Sphagnum communities in NSW and the ACT and their conservation management. Cunninghamia **7**:463–500.

Whinam J, Comfort M. 1996. The impact of commercial horse riding on sub-alpine environments at Cradle Mountain, Tasmania, Australia. Journal of Environmental Management **47**:61–70.

Whinam J, Copson G. 2006. Sphagnum moss: an indicator of climate change in the sub-Antarctic. Polar Record **42**:43–9.

Whinam J, Hope G, Good R, Wright G. 2010. Post-fire experimental trials of vegetation restoration techniques in the peatlands of Namadgi (ACT) and Kosciuszko National Parks (NSW), Australia. Pages 363–379 in S. G. Haberle, J. Stevenson, and M. Prebble, editors. Altered Ecologies: Fire, climate and human influence on terrestrial landscapes. ANU Press.

Whinam J, Hope GS, Clarkson BR, Buxton RP, Alspach PA, Adam & P, Adam P. 2003. Sphagnum in peatlands of Australasia: Their distribution, utilisation and management. Wetlands Ecology and Management **11**:37–49.

White AK. 2009. Modelling the impact of climate change on peatlands in the Bogong High Plains, Victoria.

University of Melbourne.

White C, Grose M, Corney S, Bennett J, Holz G, Sanabria L, McInnes KL, Cechet RP GS& BN. 2010. Climate futures for tasmania. Technical Report. Available from https://d1wqtxts1xzle7.cloudfront.net/39971003/Climate_Futures_for_Tasmania_extreme_eve20151113-30451-1qo115x.pdf?1447425087=&response-content-disposition=inline%3B+filename%3DClimate_Futures_for_Tasmania_extreme_eve.pdf&Expires=1608594259&Signature=bL7x4.

Williams DD, Feltmate BW. 1992. Aquatic Insects. CAB International, Wallington.

Williams KJ, Potts BM. 1996. The natural distribution of Eucalyptus species in Tasmania. Tasforests **8**:32–149.

Williams R. 1990. Cattle grazing within subalpine heathland and grassland communities on the Bogong High Plains: disturbance, regeneration and the shrub-grass balance. Proceedings of the Ecological Society of Australia **16**.

Williams R, Papst, Warwick(Williams et al. in Lindemayer et al. 2014 B, McDougall K, Mansergh I, Heinze D, Camac J, Nash M, Morgan J, Hoffmann A. 2014. Alpine ecosystems. Pages 167–212 in D. B. Lindenmayer, E. L. Burns, N. Thurgate, and A. Lowe, editors. Biodiversity and Environmental Change. CSIRO Publishing.

Williams RJ. 1987. Patterns of air temperature and accumulation of snow in subalpine heathlands and grasslands on the Bogong High Plains, Victoria. Austral Ecology **12**:153–63.

Williams RJ et al. 2015. An International Union for the Conservation of Nature Red List ecosystems risk assessment for alpine snow patch herbfields, South-Eastern Australia. Austral Ecology **40**:433–443.

Williams RJ, Ashton DH. 1987. Effects of disturbance and grazing by cattle on the dynamics of heathland and grassland communities on the bogong high plains, Victoria. Australian Journal of Botany **35**:413–431.

Williams RJ, Ashton DH. 1988. Cyclical patterns of regeneration in subalpine heathland communities on the bogong high-plains, Victoria. Australian Journal of Botany **36**:605–619.

Williams RJ, Papst WA, Wahren C-H. 1997. The Impact of Cattle Grazing on Alpine and Subalpine Plant Communities of the Bogong High. Report to the Victorian Department of Natural Resources and Environment.

Williams RJ, Wahren CH, Bradstock RA, Müller WJ. 2006. Does alpine grazing reduce blazing? A landscape test of a widely-held hypothesis. Austral Ecology **31**:925–936.

Williams RJ, Wahren CH, Tolsma AD, Sanecki GM, Papst WA, Myers BA, McDougall KL, Heinze DA, Green K. 2008. Large fires in Australian alpine landscapes: Their part in the historical fire regime and their impacts on alpine biodiversity. International Journal of Wildland Fire **17**:793–808.

Williams WD. 1964. Some chemical features of Tasmanian inland waters. Australian Journal of Marine and Freshwater Research **15**:107–122.

Williams WD, Walker KF, Brand GW. 1970. Chemical composition of some inland surface waters and lake deposits of New South Wales, Australia. Australian Journal of Marine and Freshwater Research **21**:103–116.

Wilson BR, Tulau M, Kuginis L, McInnes-Clarke S, Grover S, Milford H, Jenkins BR. 2021. Distribution, nature and threats to soils of the Australian Alps: A review. Austral Ecology **47**:166–188. Available from https://onlinelibrary.wiley.com/doi/10.1111/aec.13115 (accessed December 2, 2024).

Wimbush DJ. 1970. Hydrological Studies on Sphagnum Bogs in the Snowy Mountains, New South Wales. University of Sydney.

Wimbush DJ, Costin AB. 1983. Trends in drainage characteristics in the subalpine zone at Kosciusko. Proceedings of the Ecological Society of Australia **12**:143–154.

Wimbush DJ, Costin ABA. 1979. Trends in vegetation at Kosciusko. III Alpine range transects, 1959-1978. Australian Journal of Botany **27**:833–71.

Woodford DJ, McIntosh AR. 2013. Effects of introduced trout predation on non-diadromous galaxiid fish populations across invaded riverscapes. Department of Conservation.

Worboys G. 2003. A Brief Report on the 2003 Australian Alps Bushfires. Mountain Research and Development **23**:294–295. Available from https://doi.org/10.1659/0276-4741(2003)023[0294:ABROTA]2.0.CO;2.

Worboys GL, Good R, Spate A. 2011. Caring For Our Australian Alps: Catchments:A Climate Change Action Strategy For The Australian Alps To Conserve The Natural Condition Of The Catchments and To Help Minimise Threats To High Quality Water Yields. Available from https://theaustralianalpsnationalparks.org/wp-content/uploads/2014/01/catchmentrpt2011_summary.pdf.

Worboys GL, Pickering CM. 2002. Managing the Kosciuszko alpine area: conservation milestones and future challenges. Gold Coast, Australia.

Yospin G, Wood S, Holz A, Bowman D, Keane R, Whitlock C. 2015. Modeling vegetation mosaics in sub-alpine Tasmania under various fire regimes. Modeling Earth Systems and Environment **1**:16.

Zheng H, Chiew FHS, Post DA, Robertson DE, Charles SP, Grose MR, Potter NJ. 2024. Projections of future streamflow for Australia informed by CMIP6 and previous generations of global climate models. Journal of Hydrology **636**:131286. Elsevier B.V. Available from https://doi.org/10.1016/j.jhydrol.2024.131286.

Zomer RJ, Xu J, Trabucco A. 2023. Version 3 of the Global Aridity Index and Potential Evapotranspiration Database. Scientific Data **9**:409.

Zylstra PJ. 2018. Flammability dynamics in the Australian Alps. Austral Ecology **43**:578–591.

Zylstra PJPJ. 2006. Fire History of the Australian Alps - Prehistory to 2003. Page Fire history of the Australian Alps. Australian Alps National Parks, Australia. Department of the Environment and Water Resources. Department of the Environment and Water Resources. Available from http://www.australianalps.environment.gov.au/publications/research-reports/fire-history.html.