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2 October 2026

Department of Climate Change, Energy, the Environment and Water
Attn EPBC Monitoring
Sent via email: epbcmonitoring@environment.gov.au

To whom it may concern,

2026 Compliance Report for EPBC Approval 2005/2040 Prominent Hill Mine

This letter is written advice in accordance with Condition 2 of *Environment Protection and Biodiversity Conservation Act 1999* (Cth) (**EPBC Act**) approval 2005/2040 as amended (**EPBC Approval**) for the period of 1 July 2025 to 30 June 2026.

Declaration of accuracy

In making this declaration, I am aware that sections 490 and 491 of the EPBC Act make it an offence in certain circumstances to knowingly provide false or misleading information or documents. The offence is punishable on conviction by imprisonment or a fine, or both. I declare that all the information and documentation supporting this compliance report is true and correct in every particular. I am authorised to bind the approval holder to this declaration and that I have no knowledge of that authorisation being revoked at the time of making this declaration.

If you have any questions or require further information, please do not hesitate to contact Kara Moore.

Yours sincerely



Digitally signed by Parsons,
Jacinta
DN: cn=Parsons, Jacinta, o=BHP
Group Operations Pty Ltd,
ou=MULTI-ALLOWED
Date: 2026.10.02 17:15:11 +0930'

Jacinta Parsons
Acting Asset President Copper South Australia

Attachments:

- | | |
|--------------|--|
| Attachment A | Summary of compliance to EPBC Approval conditions (EPBC 2005/2040) |
| Attachment B | Prominent Hill Autumn Ecological Survey Report 2026 |

Attachment A Summary of compliance to EPBC Approval conditions (EPBC 2005/2040)

The current EPBC Approval conditions and evidence of compliance for this reporting period have been provided in Table 1.

Table 1: EPBC Act conditions and evidence of compliance

Condition No.	Condition	Is the project compliant with the condition	Evidence of compliance
1	To mitigate impacts to the thick-billed grass wren (<i>Amytornis modestus indulkanna</i>) and the plains rat (<i>Pseudomys australis</i>), the person taking the action must, prior to 1 July 2018, submit an Environment Management Plan (EMP) for the approval of the Minister. The EMP must be prepared in accordance with the Department's Environmental Management Plan Guidelines and include, but not be limited to: a) Management measures to maintain or improve habitat condition including: i. Measures to prevent and control human and stock access to species habitat; and ii. The control of feral predator and weed species; and b) Feral predator, weed and habitat condition monitoring, triggers for management intervention and correction actions. The person taking the action must implement the approved EMP.	Compliant	Eco Logical Australia (ELA) completed annual vegetation and habitat monitoring throughout June and August 2026 (Attachment B). The survey was delayed twice due to significant rainfall events that restricted access to monitoring sites. The monitoring meets the requirements of the approved Threatened Fauna Management Plan for Prominent Hill (OZ Minerals 2018*). ELA found no sustained significant difference between the control sites and impact sites, including mean perennial species abundance, richness and total diversity, mean recruitment scores and mean soil/ecosystem function scores. This demonstrates success in meeting the requirements of the outcome criteria for this Environmental Outcome. ELA completed weed monitoring as part of the 2026 ecological survey. ELA detected buffel grass (<i>Cenchrus ciliaris</i>) in one weed monitoring site and no significant increase in abundance of existing species. Feral animal control occurred as needed based on sighting reports. A total of seven cats were captured and euthanized during the reporting period.
2	The person taking the action must, unless otherwise agreed by the Minister, submit by 30 September of each year (beginning in 2019) written advice to the Minister demonstrating how the person taking the action has complied with the conditions of the approval.	Non-compliant	The Compliance Report was submitted to DCCEE on 2 October 2026
3	The person taking the action may choose to revise the plan approved by the Minister under condition 1 without submitting it for approval under section 143A of the EPBC Act, if the taking of the section in accordance with the revised plan would not be likely to have a new or increased impact. If the person taking the action makes this choice they must notify the Department in writing that the approved	N/A	N/A

Condition No.	Condition	Is the project compliant with the condition	Evidence of compliance
	plan has been revised and provide the Department, at least four weeks before implementing the revised plan, with: i. An electronic copy of the revised plan; ii. An explanation of the differences between the revised plan and the approved plan; and iii. The reasons the person taking the action considers that the taking of the action in accordance with the revised plan would not be likely to have a new or increased impact.		
4	The person taking the action may revoke its choice under condition 3 at any time by giving written notice to the Department. If the person taking the action revokes the choice to implement the revised plan, without approval under section 143A of the EPBC Act, the plan approved by the Minister must be implemented.	N/A	N/A
5	If the Minister give a notice to the person taking the action that the Minister is satisfied that the taking of the action in accordance with the revised plan would be likely to have a new or increased impact, then: i. Condition 3 does not apply, or ceases to apply, in relation to the revised plan, and ii. The person taking the action must implement the plan approved by the Minister. To avoid any doubt, this condition does not affect any operation of conditions 3 and 4 in the period before the day the notice is given. At the time of giving the notice, the Minister may also notify that for a specified period of time condition 3 does not apply for the plan required under the approval. Conditions 3, 4 and 5 are not intended to limit the operation of section 143A of the EPBC Act which allows the person taking the action to submit a revised plan to the Minister for approval.	N/A	N/A

*Reference: OZ Minerals (2018) *Threatened Fauna Management Plan Prominent Hill*. EPBC 2005/2040 Prominent Hill Copper-Gold Project. PH-9999-SEC-PLN-0052. Dated 15 August 2018.

Attachment B Prominent Hill Autumn Ecological Survey Report 2026



Prominent Hill ecological survey 2026

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Document Tracking

Project Name: Prominent Hill ecological survey 2026

Project Number: 600-26MEL12999

Project Manager: Ali Jeffery

Version	Prepared by	Reviewed by	Approved by	Status	Date
1	Ali Jeffery	Dr Meredith Henderson	Jasmine Richards	Draft	03/09/2026
2	Ali Jeffery	Jasmine Richards	Jasmine Richards	Final	17/09/2026

This report should be cited as 'Eco Logical Australia 2026, Prominent Hill ecological survey 2026, Prepared for BHP.'

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Disclaimer

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Abbreviations

Abbreviation	Description
BOM	Bureau of Meteorology
DCCEEW	Department of Climate Change, Energy, the Environment and Water
DEM	Department for Energy and Mining
ELA	Eco Logical Australia
EML	Extractive Mineral Leases
EPBC Act	<i>Environment Protection and Biodiversity Act 1999</i> (Commonwealth)
GPS	Global Positioning System
IBRA	The Interim Biogeographic Regionalisation for Australia
LIC	Leading Indicator Criteria
Mining Act	<i>Mining Act 1971</i> (SA)
ML	Mining Lease
MPL	Miscellaneous Purpose Licences
NEPA	National Environment Protection Authority
NVC	Native Vegetation Council
OMC	Outcome Measurement Criteria
PEPR	Program for Environment Protection and Rehabilitation
PH01 – PH27	Monitoring sites
SAAL	South Australian Arid Lands
SE	Standard Error
SEB	Significant Environmental Benefit

Executive Summary

Eco Logical Australia Pty Ltd (ELA) was engaged by BHP (formerly OZ Minerals) to undertake the 2026 ecological compliance monitoring survey at the Prominent Hill copper and gold mine. Prominent Hill (Mining Lease (ML) 6228 and associated tenements) is located approximately 650 km north-west of Adelaide and 136 km south-east of Coober Pedy, South Australia.

The 2026 survey represents the tenth year of monitoring since the survey methodology was updated under the 2017 Program for Environment Protection and Rehabilitation (PEPR). BHP has a legislative requirement under the approved PEPR to conduct ecological monitoring on ML 6228, within the surrounding Significant Environmental Benefit (SEB) offset area and at several weed sites along main access routes, to assess the abundance and diversity of native flora at control and impact sites.

Twenty permanent floristic monitoring quadrats have been established (10 impact and 10 control), positioned to detect potential impacts of the mine by comparing sites within or close to the ML area (impact) with sites in the surrounding Significant Environmental Benefit (SEB) offset area (control). The quadrats are located within three vegetation associations: Acacia Woodland, Chenopod Shrubland and Mallee Woodland, with Chenopod Shrubland being the dominant vegetation type. Weed monitoring sites are located along the Access Road to the Stuart Highway and the Haul Road to the Wirrida Rail Siding to assess the potential for weed introduction as a result of the Prominent Hill operation.

The field survey for the Autumn 2026 ecological monitoring was undertaken between 18 – 22 May 2026. This survey completed monitoring at four weed monitoring sites (1,2, 3 and 4) and one floristic monitoring quadrat (PH22), however was then postponed due to rain. Therefore, a second field survey was undertaken from 3 – 5 August 2026 to complete the assessment.

In total 17 of the 20 floristic monitoring quadrats, Jessup transects and subplots were able to be completed across these two survey periods. Three of the 20 permanent floristic monitoring quadrats were inaccessible during the survey period being:

- PH07 Mallee Woodland (Control)
- PH11 – Chenopod Shrubland (Control)
- PH20 Chenopod Shrubland (Control).

The 2026 ecological survey recorded 148 flora species (145 native and three exotic) across the collected from the vegetation quadrat monitoring sites. The results of the statistical analysis found that total native species richness in sites supporting Chenopod Shrubland vegetation was significantly different between impact and control sites. The difference in species richness is likely due to pre-existing differences between control and impact sites and vegetation variation and fluctuations between survey years. Regardless, this result should be monitored in future surveys. Other than this, statistical analysis did not indicate significant differences between control and impact sites for species richness, perennial richness, vegetation cover, perennial cover, perennial density or recruitment in 2026. The results indicate that the monitored rangeland system remains stable in the vegetation types assessed; Acacia Woodland, Chenopod Shrubland and Mallee Woodland. Native species abundance, richness, diversity and recruitment within impact sites do not indicate trends different from those measured within the control sites, across vegetation types.

Weed monitoring identified *Cenchrus ciliaris* (buffel grass) at weed monitoring site 4, which is the first record of this species in the since the ecological compliance monitoring survey began in 2017. *Malvastrum americanum* var. *americanum* was recorded at weed monitoring sites 3 and 4 and was observed in 15 of the 17 floristic monitoring quadrats. *Sonchus oleraceus* was identified within 3 of 17

floristic monitoring quadrats. These records indicate that weed species remain present within the monitoring area and should continue to be monitored closely, alongside ongoing management actions.

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1. Introduction

1.1. Site context

The Prominent Hill mine site is a copper and gold mine located approximately 650 km north-west of Adelaide, South Australia (SA). The Prominent Hill Mine is owned and operated by BHP Group Limited.

The Mining Lease (ML) for Prominent Hill (ML 6228) was approved in 2006 for production of copper and gold. The Prominent Hill operation is comprised of a number of tenements approved under the *Mining Act 1971* (Mining Act) between 2006 and 2010, including ML 6288, Miscellaneous Purpose Licences (MPL 81, 82, 83,84, 91, 93, 94 96, 97, 101, 112-117 and 119-122) and Extractive Mineral Leases (EML 6234, 6236-6242, 6278-6296, 6299-6301).

Based on the Interim Biogeographic Regionalisation for Australia (IBRA), Prominent Hill mine is situated on the Stony Plains Rangeland bioregion (Oodnadatta subregion STP02), and within the Marla-Oodnadatta district that is characterised by gibber tablelands, flood out plains, dunes systems and breakaway country. The ML 6228 is characterised by low opened Chenopod Shrublands, and low open woodlands on stony watercourses and outwash plains.

1.2. Background and purpose of this report

Monitoring at Prominent Hill has been occurring since 2005. The monitoring previously assessed native vegetation communities and fauna presence with a focus on threatened species within the landscape.

In 2017, Jacobs completed a review to determine whether the ecological monitoring program was fit for purpose to meet the lease conditions and Program for Environment Protection and Rehabilitation (PEPR) outcomes. As a result of this review, site numbers were reduced from 34 to 20, seven of which were new sites, and, as a surrogate for fauna surveys, flora sites were selected based on fauna habitat condition. Due to these methodology changes, and difficulty locating previous quadrats by reason of lack of physical quadrat set up and GPS information, comparisons with years prior to 2017 is problematic. Therefore, data collected in 2017 is now used as baseline data for the floristic monitoring at Prominent Hill in support of the PEPR.

The following report is based on compliance requirements as outlined in the PEPR. This floristic monitoring report will seek to demonstrate compliance with PEPR outcome measurement criteria (OMC). ELA understands this monitoring report will be made available to the Department for Energy and Mining (DEM), who assess BHP's ongoing compliance with vegetation and habitat conditions by review of performance against the OMC (including leading indicator criteria (LIC)) detailed in Table 1. It is understood that it will also be provided to the Department of Climate Change, Energy, the Environment and Water (DCCEEW)(now National Environment Protection Authority (NEPA)) regarding Prominent Hills *Environment Protection and Biodiversity Act 1999* (EPBC Act) conditions, however this is not discussed further within the document.

1.3. Objectives

The objectives of the 2026 Prominent Hill annual ecological assessment were to:

- Undertake monitoring at 20 permanent floristic monitoring quadrats to assess potential changes in species diversity, vegetation cover, recruitment, perennial plant densities, the percentage of ground cover (grass, litter), bare ground and any changes to the abundance or diversity of native vegetation or a reduction in habitat quality at both impact and control sites (see Figure 1).

- Undertake weed monitoring at five permanent weed monitoring sites to determine whether there has been introduction of new weed species or an increase in abundance or distribution of existing species (see Figure 1).
- Prepare an ecological assessment report that presents the findings of the assessment and considers potential impacts of the Prominent Hill mine on vegetation and habitat condition.

It is noted that in 2026, three of the 20 permanent floristic monitoring quadrats were inaccessible during the survey period. These quadrats were:

- PH07 Mallee Woodland (Control)
- PH11 – Chenopod Shrubland (Control)
- PH20 – Chenopod Shrubland (Control).

Therefore, the total numbers for the floristic monitoring quadrats, Jessup transects and subplots for the vegetation monitoring in 2026 have been reduced from 20 to 17.

1.4. Scope

To meet above listed objectives, ELA has undertaken a survey comprising the following aspects (as contained within the PEPR and the *what will be measured* column of Table 1 habitat quality indicators including vegetation and soils (representing broader ecosystem function)):

- 17 floristic monitoring quadrats (100 m x 100 m) (Biological Survey Method (DENR 1997))
 - panoramic photographs collected to aid in identifying temporal change in vegetation cover and structure
 - identification of all aboveground vascular flora species present (species diversity/ richness, inclusive of annuals)
 - cover (%) of individual species
 - species recruitment and life form present.
- 17 Jessup Transects (100 m x 4 m)
 - separate counts of adult and juvenile long-lived perennials (woody shrubs)
 - stem counts for perennial species
 - counts of reproductive individuals for perennial species.
- 17 subplots (10 x 2 m) within the vegetation quadrat, for each 1 x 1 m unit of sub-plot:
 - estimate of % grass cover (ephemerals, annuals)
 - estimate of % bare ground
 - estimate of % litter cover within the plot
 - estimate of % surface crust
 - counts of recruits (all shrubs) to provide recruitment score
 - long lived perennials (over and under storey) via species abundance counts (density) for both juveniles and adults.
- observational data including vehicle tracks, erosion, vegetation clearing, dust, inappropriate access, feral animals (and weeds), counts and identification of any declared weed infestations observed along transects at each location.

Table 1: Summary of outcome measurement criteria for flora at Prominent Hill (Reference: Table 7-12 in the 2022 PEPR)

Impact event	Environmental outcome	Outcome measurement criteria (OMC) details					Leading indicator (LI) criteria
		What will be measured and form (method) of measurement	Location	Outcome achievement	Frequency	Control or baseline data	
Reduced conditions favourable for plant growth due to disturbance, dust, salinity, altered surface water flow or soil erosion. Reduced species abundance due to vegetation clearing, increased grazing resulting from increased water. Significant impacts to threatened species due to vegetation clearing. Altered vegetation composition due to increase in fire ignition and/or changes in fire frequency and intensity.	Environmental offsets are approved and in place for all clearance of native vegetation	Habitat Quality Indicators including vegetation and soils (representing broader ecosystem function) 100 x 100 m vegetation quadrat (Biol. Survey Method) - identification of all species present (species diversity/richness, inclusive of annuals) - cover (%) of individual species - species identified as recruiting. 10 x 2m subplot within vegetation quadrat. For each 1 x 1m unit of sub-plot: - estimate of % grass cover (ephemerals, annuals) - estimate of % bare ground - estimate of % litter cover within the plot - estimate of % surface crust - counts of recruits (all shrubs) to provide recruitment score - long lived perennials (over and under storey) via species abundance counts (density) for both juveniles and adults. Panoramic photographs collected to aid in assessment of vegetation cover for recruitment. Observational data to be collected at all sites including (but not limited to) vehicle tracks, erosion, vegetation clearing, distance to mine site, light, dust, inappropriate access, feral animals and weeds. GIS output of approved clearance boundary and actual clearance boundary.	See Figure 7-6 (in PEPR 2022). Assessment will be undertaken at a series of permanent impact sites (adjacent to mining operations) and control sites (replicate sites remote from mining operations but within SEB) to enable comparison between potential mine related activities and seasonal variations. Sites selected in representative habitats for key fauna species, including EPBC and NPW listed species, and to be representative of the broad vegetation community types present within the ML, MPL and SEB areas. Impact sites (PH01, PH02, PH03a, PH10, PH12, PH 13a, PH 15, PH19, PH22, PH23) and control sites (PH06, PH07, PH11, PH16, PH17, PH18, PH20, PH24, PH25, PH27).	Approved Native Vegetation Management Plan and SEB Offset in place.	Annually	REP021 Prominent Hill Analysis of Flora and Fauna Monitoring Data 2007 to Spring 2012. Other Flora survey reports, conducted at the site biannually between 2007 and 2016. Comparable data up until methodology changes in 2015. Historical information can be found in Jacobs (2017) - 2017 Review of Prominent Hill Ecological Monitoring. All annual ecological monitoring reports.	Reduction of perennial species abundance (counts) at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods. Suppression of recruitment indicated by a reduction in recruitment index scores at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods. An increase in bare ground and/ or scald/ erosion % cover at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods. If leading indicator triggered, further assessment of detected impacts vs pre-mine condition by comparison with sites outside of SEB area required to determine non-compliance with lease condition. Annual review of vegetation clearance confirms all clearance has been approved
Reduced species abundance due to vegetation clearing, increased grazing from increased surface water, traffic, noise and vibration and downstream effects. Significant impacts to threatened species due to vegetation clearing and predation	No loss of abundance or diversity of native vegetation, or reduction in habitat quality, on or off the Mining and or miscellaneous purposes lease areas during construction, operation as a result of mining activities unless prior approval under relevant legislation is obtained and environmental offsets are approved and in place.	As above	As above	As above	As above	As above	Indicators of habitat degradation, including: Reduction of perennial species abundance (counts) at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods. Suppression of recruitment indicated by a reduction in recruitment index scores at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods.

Impact event	Environmental outcome	Outcome measurement criteria (OMC) details					Leading indicator (LI) criteria
		What will be measured and form (method) of measurement	Location	Outcome achievement	Frequency	Control or baseline data	
							An increase in bare ground and/or scald/erosion % cover at impact sites without a corresponding increase at control sites over three consecutive monitoring periods. If leading indicator triggered, implement targeted threatened bird surveys to confirm ongoing presence of thick-billed grasswren and chestnut-breasted whiteface within impacted sites.
Increased weed density and distribution as a result of Prominent Hill operations Introduction of new weed species as a result of Prominent Hill operations	No introduction of new species of weeds, plant pathogens or pests (including feral animals), nor sustained increase in abundance of existing weed or pest species in the licence area compared to adjoining land as a result of mining operations.	Records of counts and identification of any declared weed infestations observed along transects at each location. Records of weed control actions implemented (where relevant). Records of feral animal observations, trapping and capture rates.	Five weed monitoring locations	Weed infestations are recorded, treated and monitored for ongoing management requirements.	Annually	PH-ENV-REP-0500 Stage Two Significant Environmental Benefit Offset Area Management Plan REPO21 Prominent Hill Analysis of Flora and Fauna Monitoring Data 2007 to Spring 2012. Other Flora and Fauna survey reports, conducted at the site biannually between 2007 and 2016. All annual ecological monitoring reports.	Annual review of the weed survey and management register (results of field monitoring and visual observations) considering trends that could indicate population increase or new weed species.
			Various locations including landfill, camp and as per opportunistic sightings	Pest animal sightings are recorded and will result in the initiation of a trapping/baiting program and subsequent monitoring.	As required		Quarterly review of cat sightings and trapping register considering trends that could indicate population increase and requirement for increase in trapping program

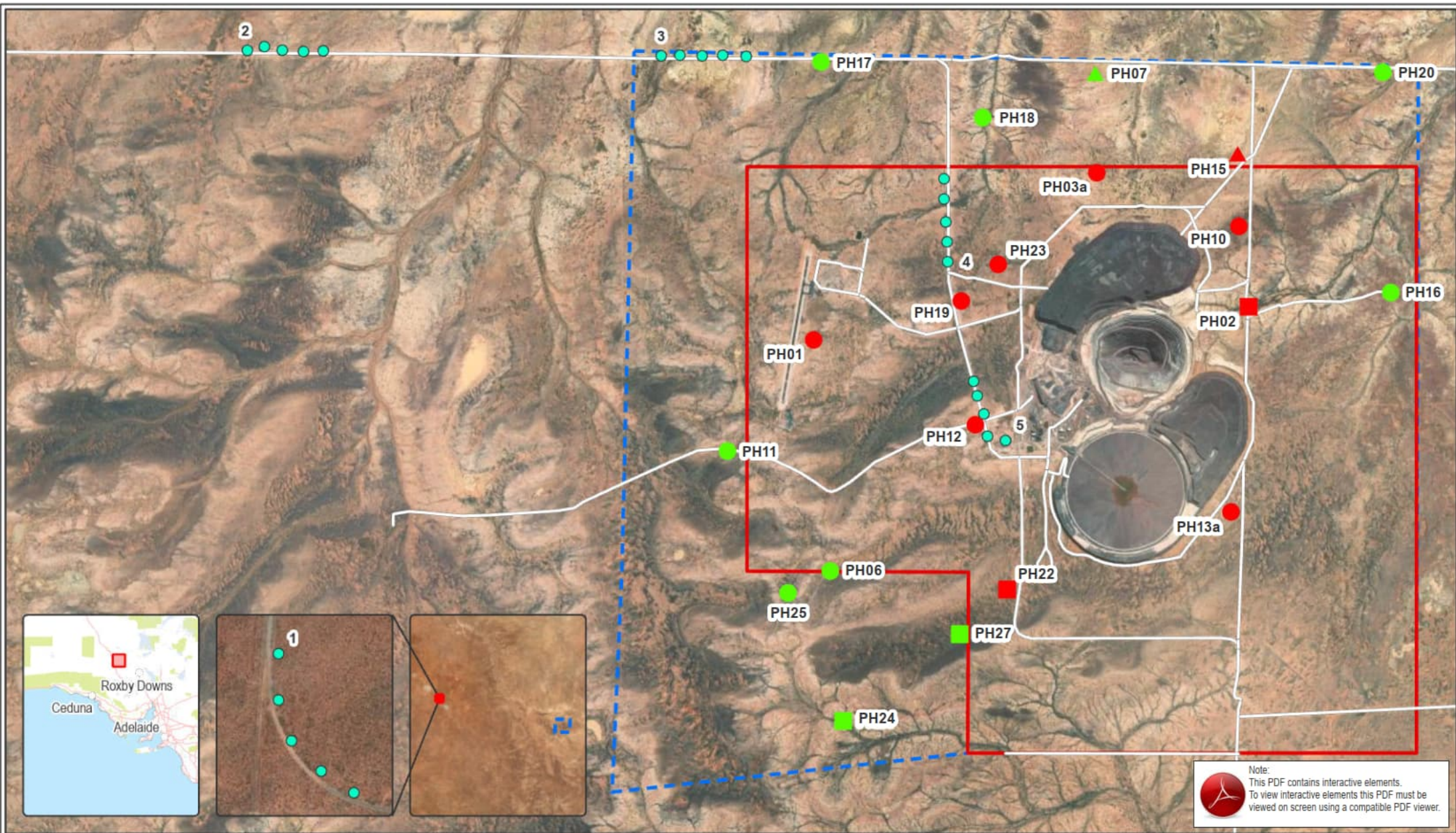


Figure 1: Prominent Hill Floristic Monitoring

- Prominent Hill mining lease area
- Significant Environmental Benefit (SEB) Offset Area
- Vegetation monitoring sites - Floristic quadrat

- Subplot
- 4m Jessup buffer
- NW corner / panorama photo point
- > 100m transect

- Weed monitoring sites
- Mine access road

Impact sites

- Acacia Woodland Impact
- Chenopod Shrubland Impact
- ▲ Mallee Woodland Impact

Control sites

- Acacia Woodland Control
- Chenopod Shrubland Control
- ▲ Mallee Woodland Control

0 1 2
Kilometres

Datum/Projection:
GDA 1994 MGA Zone 53
24ADL7673-OK Date: 6/12/2025



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2. Methodology

The methodology outlined in the sections below is consistent with the methodology of the PEPR (2022) (Table 1).

2.1. Survey team

The field survey for the Autumn 2026 ecological monitoring was undertaken between 18 – 22 May 2026 by ELA Ecologists Ali Jeffery and Georgia Wilson. This survey completed monitoring at four weed monitoring sites (1,2, 3 and 4) and one vegetation monitoring quadrat (PH22), however was postponed due to rain. Therefore, a second field survey was undertaken from 3 – 5 August 2026 by ELA Ecologists Ali Jeffery and Georgia Wilson to complete the assessment.

2.2. Habitat quality indicators

Habitat quality indicators are outlined as per the PEPR.

2.2.1. Monitoring locations

A total of 17 permanent floristic monitoring quadrats, comprising 10 control sites and 7 impact sites, were surveyed in 2026 to measure habitat quality indicators against the PEPR (2022) OMC (Table 2 and Figure 2). Each quadrat covers 1 ha (100 m x 100 m) and contains a 10 m x 2 m subplot located in the north-west corner, as well as a 100 m Jessup transect with a 4 m investigational buffer running from the west to the east boundary of the quadrat (Figure 2).

The locations of the quadrats were selected in 2017 to provide a representative sample of the three dominant vegetation types within the Prominent Hill ML area: Acacia Woodland, Chenopod Shrubland and Mallee Woodland. The corners of each quadrat and subplot, together with the start and end points of each Jessup transect, are permanently marked with metal star pickets to ensure consistent data collection between monitoring years (Figure 2).

In 2026, three of the 20 established permanent floristic monitoring quadrats were inaccessible during the survey period: PH07 Mallee Woodland (Control), PH11 Chenopod Shrubland (Control) and PH20 – Chenopod Shrubland (Control). As a result, vegetation quadrat, Jessup transect and subplot data were collected from 17 sites only. The surveyed quadrats and their associated vegetation communities are listed in Table 2.

Table 2: Floristic monitoring quadrats by vegetation type and treatment (impact/ control)

Vegetation type	Impact or Control	Floristic monitoring quadrats
Chenopod Shrubland	Control	PH06, PH11 , PH16, PH17, PH18, PH20 , PH25
Chenopod Shrubland	Impact	PH01, PH03, PH10, PH12, PH13, PH19, PH23
Acacia Woodland	Control	PH24, PH27
Acacia Woodland	Impact	PH02, PH22
Mallee Woodland	Control	PH07
Mallee Woodland	Impact	PH15

*Floristic monitoring quadrats inaccessible during the 2026 survey are presented with a ~~strikethrough~~

2.2.2. Monitoring methods

Each floristic monitoring quadrat comprised three components: a vegetation quadrat, a Jessup transect and a subplot. The latitude and longitude of each quadrat corner were recorded, and a panoramic photograph was taken from the north-west corner of the quadrat (Figure 2). Each vegetation monitoring quadrat was surveyed using a meandering search across the entire 1 ha area and the following data was collected:

- identification of all species present (species diversity/ richness, inclusive of annuals)
- cover (%) cover of individual species.
- species identified as recruiting.
- record of life-forms present for each species.

A fixed subplot (2 m × 10 m) located in the north-west corner of each quadrat, was used to estimate ground cover attributes and count perennial recruits. These measurements included:

- estimate of % grass cover (ephemerals, annuals)
- estimate of % bare ground
- estimate of % litter cover within the plot
- estimate of % surface crust
- counts of recruits (all shrubs) to provide recruitment score.

A Jessup transect, incorporating an investigational buffer belt transect (4 m × 100 m), was positioned in the centre of each quadrat in accordance with the survey methodology outlined in the Pastoral Lease Assessment Technical Training Manual 2011 (DENR 2011). This includes:

- individual counts for mature and juvenile perennial shrubs
- individual stem counts for mature and juvenile perennial shrubs

The latitude and longitude of each corner point of the vegetation monitoring quadrat, as well as the start and end points of the Jessup transect, were recorded.

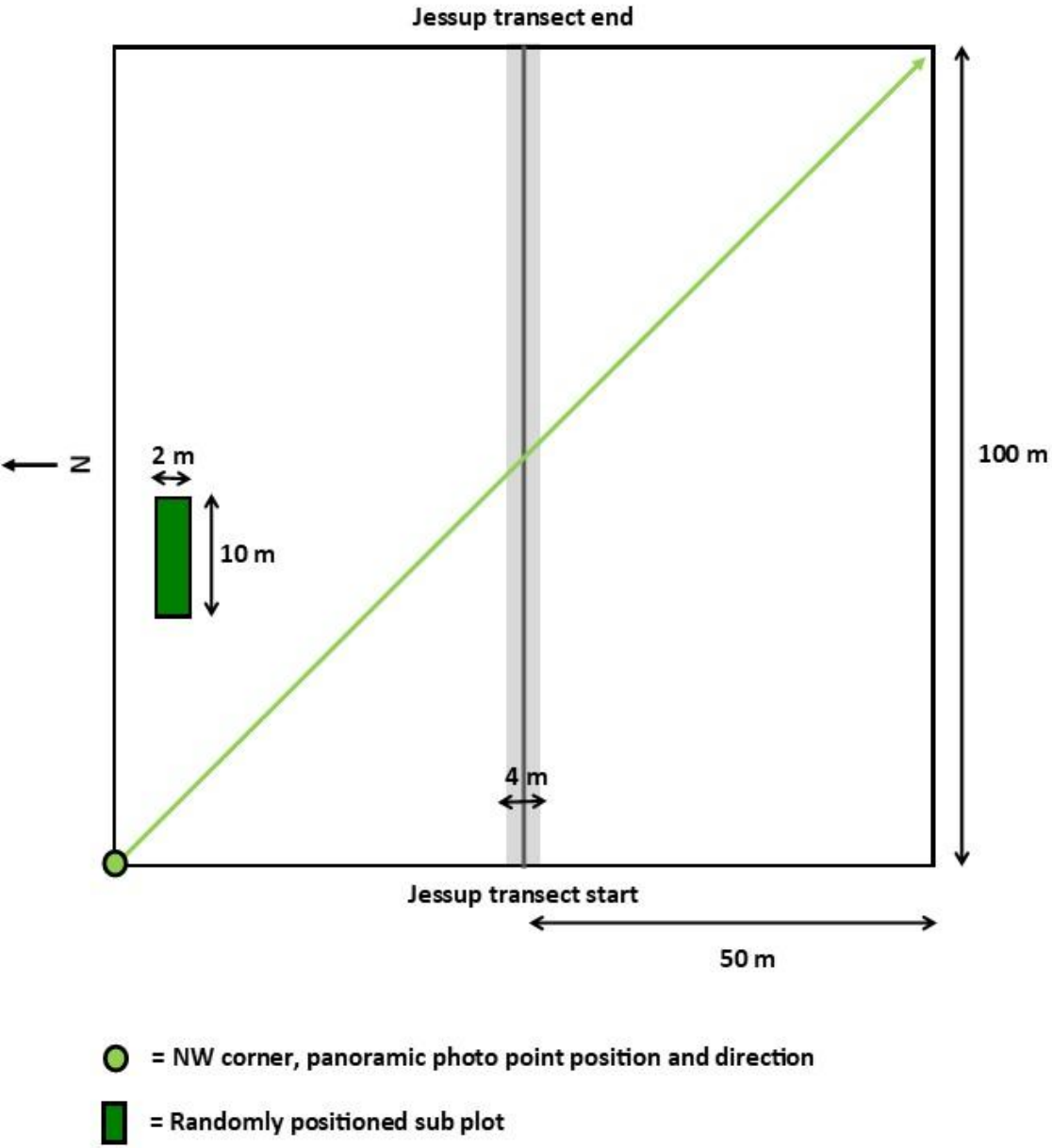


Figure 2: Floristic monitoring quadrat diagram

2.2.3. Statistical analysis

2.2.3.1. Species diversity

Mean total species richness, defined as the number of flora species recorded, was calculated for impact and control sites within each vegetation type: Acacia Woodland, Chenopod Shrubland and Mallee Woodland. Differences in total species richness between impact and control sites were assessed separately for each vegetation type using two-sided independent-samples Welch *t*-tests. The Welch *t*-test was selected because it does not assume equal variances between the impact and control groups.

The same approach was used to compare mean perennial species richness between impact and control sites within each vegetation type. Statistical significance was assessed using a threshold of $p \leq 0.05$. Where $p > 0.05$, the null hypothesis of no difference in mean species richness between impact and control sites was not rejected. Analyses were conducted only for vegetation types where both impact and control sites were represented.

These analyses address the LIC outlined in the PEPR (Table 1), specifically regarding whether there was any indication of:

‘Reduction of perennial species abundance (counts) at impact sites without a corresponding reduction at control sites.’

2.2.3.2. Native vegetation cover

Mean vegetation cover, expressed as percentage cover, was calculated for impact and control sites within each vegetation type: Acacia Woodland, Chenopod Shrubland and Mallee Woodland. Differences in total vegetation cover between impact and control sites were assessed separately for each vegetation type using two-sided independent-samples Welch *t*-tests. The Welch *t*-test was selected because it does not assume equal variances between the impact and control groups.

This was used to assess whether BHP are trending towards achievement of the environmental outcome outlined in the PEPR (Table 1):

‘No loss of abundance or diversity of native vegetation, or reduction in habitat quality on or off the Mining and or miscellaneous purposes lease areas during construction, operation as a result of mining activities.’

2.2.3.3. Perennial density

Perennial density (stems/ m²) was recorded for mature and juvenile perennial plants using a Jessup belt transect (Figure 2) for each vegetation type and treatment combination. Total perennial stem density, mature perennial density and juvenile perennial density were calculated separately. Differences between impact and control sites were assessed independently within each vegetation type using two-sided independent-samples Welch *t*-tests. The Welch *t*-test was used because it does not assume equal variances between impact and control groups. Statistical significance was assessed using a threshold of $p \leq 0.05$. Where $p > 0.05$, the null hypothesis of no difference in mean perennial density between impact and control sites was not rejected. Analyses were conducted for vegetation types where both impact and control sites were represented.

Statistical analyses were undertaken to address the following LIC:

‘Reduction of perennial species abundance (counts) at impact sites without a corresponding reduction at control sites.’

'Reduction of perennial species abundance (counts) at impact sites without a corresponding reduction at control sites.';

And;

'Suppression of recruitment indicated by a reduction in recruitment index scores at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods.'

2.2.3.4. Native recruitment

Native recruitment, expressed as the proportion of native perennial species present that had recruits, was determined for each treatment and vegetation type combination. Differences in the proportion of recruiting native perennial species between impact and control sites were assessed independently for each vegetation type using two-sided independent-samples Welch *t*-tests. The Welch *t*-test was used because it does not assume equal variances between impact and control groups. Statistical significance was assessed using a threshold of $p \leq 0.05$. Where $p > 0.05$, the null hypothesis of no difference in mean native recruitment between impact and control sites was not rejected. Analyses were conducted for vegetation types where both impact and control sites were represented.

These analyses address the LIC outlined in the PEPR (Table 1), specifically regarding whether there was any indication of:

'Suppression of recruitment indicated by a reduction in recruitment index scores at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods.'

2.2.3.5. Subplot monitoring

The ground cover metrics measured in the subplots include grass cover (%), bare ground (%), litter cover (%), soil crust (%) and scalded surface/ erosion (%). Maintaining ground cover is paramount in the South Australian Arid Lands (SAAL) as it helps to protect soils from erosion as well as providing protection for seed germination (DPI 2006).

Mean values for each ground-cover metric—grass cover, bare-ground cover, litter cover, soil-crust cover and scalded-surface/ erosion cover—were calculated for impact and control sites within each vegetation type. Differences in groundcover estimates between impact and control sites were assessed separately for each vegetation type using two-sided independent-samples Welch *t*-tests. The Welch *t*-test was selected because it does not assume equal variances between impact and control groups. Statistical significance was assessed using a threshold of $p \leq 0.05$. Where $p > 0.05$, the null hypothesis of no difference between impact and control sites was not rejected. Analyses were conducted only for vegetation types where both impact and control sites were represented.

Analyses of ground cover were undertaken to address a specific LIC in the PEPR (Table 1) regarding whether there is any indication of

'An increase in bare ground and/or scald erosion % cover at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods.',

'Reduction of perennial species abundance (counts) at impact sites without a corresponding reduction at control sites.';

And;

'Suppression of recruitment indicated by a reduction in recruitment index scores at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods.'

2.3. Weed monitoring transects

In addition to the floristic monitoring quadrats, five permanent weed monitoring transects were also surveyed during the 2026 survey period. The permanent weed monitoring transects were established in 2017 and are monitored as a requirement of the OMC detailed within the PEPR (Table 1).

The five-weed monitoring transects are located alongside the mine Access Road towards the Stuart Highway and the Haul Road to the Wirrida Rail Siding, located approximately 198 km (north-west) from the mine, (Figure 1). Each of the five weed monitoring sites comprise five consecutive 100 m x 50 m transects spaced 250 m apart. The weed monitoring sites were traversed on foot, and data collected includes weed species and abundance.

2.4. Weather data

The Stony Plains Bioregion experiences an arid climate with extreme temperatures and a spatially averaged annual rainfall of 118 mm for the period 1890–2005 (Commonwealth of Australia 2008).

ELA used rainfall data from 2009–2026 supplied by BHP from the Prominent Hill weather station to assess local rainfall patterns and establish a long-term rainfall average for the monitoring period. Rainfall at Prominent Hill remains highly variable between months and seasons, with the highest mean monthly rainfall recorded in January (21.7 mm), followed by February (18.6 mm) and March (17.6 mm). Seasonal rainfall was greatest in summer (18.2 mm), followed by autumn (14.7 mm), spring (12.7 mm) and winter (7.5 mm).

Rainfall in early 2026 was above average in February (85.2 mm), March (60.6 mm) and May (24.2mm). Winter months also recorded relatively high average rainfalls, with 17.8 mm was recorded in June, and 19.4 mm in August. The two field visits in 18 – 22 May and 3 – 5 August, were conducted after relatively wet period, which may have favoured annual species emergence and short-term growth responses. These rainfall conditions are likely to have influenced floristic observations, particularly the abundance of annuals, species richness and recruitment at the time of survey.

Table 3: Historical rainfall data Prominent Hill weather station data for November 2009 to August 2026

Monthly rainfall (mm)													
Year	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Annual
2009	12.4											30.6	-
2010	9.2	4.6	0.4	10	42.8	26.4	0	2.8	27.2	24.8	22.2	29.8	200.2
2011	23.2	15.6	64.6	62.0	0.0	20.2	0.8	13.0	1.4	0.6	20.4	8.0	229.8
2012	18.8	4.0	3.4	49.0	5.8	0.0	2.8	0.4	0.0	0.0	0.6	5.0	89.8
2013	2.8	0.0	4.8	15.0	2.6	36.0	36.6	4.8	0.0	0.6	2.0	1.0	106.2
2014	2.6	1.2	14.2	1.6	75.2	8.0	0.8	1.0	8.6	1.6	0.8	5.2	120.8
2015	24.8	43.4	0.0	0.8	1.6	37.4	8.0	0.6	1.8	0.2	0.2	2.2	121
2016	42.2	5.4	5.0	41.6	0.2	22.2	20.8	5.6	25.4	23.2	1.4	1.6	194.6
2017	12.6	84.2	2.0	0.0	24.6	4.6	9.0	0.0	0.6	0.2	8.0	41.0	186.8
2018	0.8	23.6	3.6	0.0	0.0	7.0	1.8	0.0	1.8	0.0	6.0	10.0	54.6
2019	0.0	6.8	0.0	0.4	5.6	0.4	2.4	0.0	0.0	2.6	0.6	2.6	21.4
2020	13.4	51.2	106.2	2.8	8.2	0.2	0.4	0.0	24.4	49.0	41.0	0.4	297.2
2021	6.4	8.4	20.0	31.8	0.0	6.4	7.6	7.0	0.0	0.0	4.4	61.8	153.8
2022	0.8	56.4	4.2	0.2	36.8	17.6	2.4	1.2	4.8	1.0	107.4	8.8	241.6
2023	60	47.4	2.8	2.0	31.2	0	42.6	0	1.6	0	0.4	12.2	200.2

Monthly rainfall (mm)													
2024	9.6	16	0	5.8	0	2.8	9.2	22.4	8.4	4	2.4	31.4	112.0
2025	3.6	0	0.2	16.2	0	0	4.2	15.2	9.2	0.6	37.6	9.4	90.2
2026	-	0.2	85.2	60.6	0	24.2	17.8	5.4	19.4	-	-	-	
Monthly mean	14.3	21.7	18.6	17.6	13.8	12.6	9.8	4.7	7.9	6.8	16	15.4	
Seasonal mean	Summer			Autumn			Winter			Spring			
	18.2			14.7			7.5			12.7			

3. Results

Results are presented as the mean $\pm$ standard error (SE), where applicable. As Mallee Woodland only has one impact and no control sites, no SE is presented.

Jessup transects were re-instated in 2020, giving four years of baseline metrics to compare the 2026 results. In the results below, metric data has been presented for analysis to include both perennial and annual species as well as perennial species alone.

As part of the analysis, independent-samples Welch t-tests were conducted to determine whether the means of species richness, vegetation cover, perennial density, recruitment and ground cover, differed significantly between control and impact sites. A significance threshold of $p = 0.05$ was used to assess the null hypothesis that there was no difference in the means between the two groups. Where the resulting p -value was less than or equal to 0.05, the null hypothesis was rejected, indicating evidence of a statistically significant difference. Where the p -value was greater than 0.05, the null hypothesis could not be rejected, indicating that the observed difference was not statistically significant and may reflect natural variation or sampling variability.

3.1. Floristic quadrat monitoring

3.1.1. Species Richness

3.1.1.1. Summary

In 2026, 148 species were recorded, comprising 145 native species and 3 exotic species. The total species richness is fewer species than recorded in the 2025 survey (161 species), and the 2024 survey (153 species). Total species richness declined between 2017 and 2019, with 116 species recorded in 2017, 93 in 2018, and 75 in 2019 (ELA 2017; ELA 2018; GHD 2019; ELA 2021), following a period of low rainfall. Species richness increased from 2020 to 2026 relative to the 2017–2019 period, largely due to the greater representation of annual species following higher rainfall in those years. In particular, 49 annual species were recorded in 2021 (ELA 2021), 30 in 2022, 39 in 2023, 39 in 2024, 34 in 2025 and 41 in 2026. This compares with zero annual species recorded in both 2018 and 2019 (ELA 2018; GHD 2019).

In 2026, 12 of the 17 floristic monitoring quadrats recorded at least one species that had not previously been identified within the vegetation monitoring sites (Table 5). However, all these species had previously been recorded within the ML area.

Table 4: New flora species records for each floristic monitoring quadrat

Flora species	Common name	Floristic monitoring quadrat(s)
<i>Acacia ligulata</i>	umbrella wattle	PH27
<i>*Carrichtera annua</i>	Ward's weed	PH22
<i>Cucumis myriocarpus</i>	paddy melon	PH02, PH13, PH16
<i>Eriochloa australiensis</i>	Australian cup grass	PH22
<i>Gunniopsis papillata</i>	twin-leaved pigface	PH24
<i>Lemooria burkittii</i>	wires and wool	PH03
<i>Maireana ovata</i>	-	PH15
<i>Polycalymma stuartii</i>	poached-egg daisy	PH13
<i>Rumex vesicarius</i>	ruby dock	PH10, PH22, PH23
<i>Senecio glossanthus</i>	slender groundsel	PH02, PH13, PH16, PH18, PH24, PH27

Flora species	Common name	Floristic monitoring quadrat(s)
<i>Sisymbrium erysimoides</i>	smooth mustard	PH03, PH23
<i>Stenopelaum sp.</i>	No widely used common name	PH03, PH23, PH25

*denotes exotic flora species

3.1.1.2. Native species richness site comparison

In 2026, data analysis of native species richness within sites supporting Acacia Woodland found mean total species richness within impact sites was 6.7% greater than at control sites (Figure 3). Mean species richness at Acacia Woodland impact sites was 47.5 species, which is the highest mean since this survey method was implemented in 2017. Acacia Woodland impact sites had a slightly higher mean total species richness than control sites (47.5 compared with 44.5 species). However, the Welch independent-samples t-test indicated that this difference was not statistically significant ($t = -0.26$, $df = 1.77$, $p = 0.822$). The 95% confidence interval for the difference in mean species richness ranged from -59.31 to 53.31 species, indicating uncertainty around the estimated difference, and given the very small sample size, these results should be interpreted cautiously.

Data analysis of native species richness within sites supporting Chenopod Shrubland found mean species richness within control sites was 23.3% greater than at impact sites (Figure 3). Mean species richness at control sites supporting Chenopod Shrubland was 47.2 species, which is the highest since this survey method was implemented in 2017. Despite control sites having a higher mean total species richness than impact sites (47.2 compared with 38.3 species), the Welch independent-samples t-test did not indicate a statistically significant difference in total species richness between Chenopod Shrubland control and impact sites in 2026 ($t = 1.98$, $df = 8.56$, $p = 0.081$). The 95% confidence interval for the difference in means ranged from -1.36 to 19.19 species, indicating that the observed difference was not statistically significant at the 0.05 level.

In 2026 there was only data from one impact site that supported Mallee Woodland, which recorded 48 species (Figure 3). No statistical analysis was conducted as there was no control data collected in 2026.

3.1.1.3. Native species richness comparison between years

In Acacia Woodland impact sites native species richness declined from 2017-2019. From 2020-2026, an overall increasing trend has been observed. In Acacia Woodland control sites, a declining trend in species richness is seen from 2017-2019, however from 2020-2026, species richness has remained relatively consistent, with consideration to seasonal variation between survey seasons.

In Chenopod Shrubland impact sites native species richness declined from 2017-2019, however from 2020-2026 species richness has remained relatively consistent, with consideration to seasonal variation between survey seasons. In Chenopod Shrubland control sites, a declining trend is seen from 2017-2019, however from 2020-2026 there has been an overall increasing trend in species richness.

In Mallee Woodland impact sites native species richness was stable from 2017-2019, with ranges between 20-25 species. From 2020-2026 native species richness has been relatively consistent, and species diversity has increased, with ranges between 40-48 species during this period. In Mallee Woodland control sites native species richness was relatively consistent from 2017 to 2023, with the exception of the 2021 survey which recorded 32 species, which is the highest recorded. Species richness was stable in 2024 and 2025, with 32 species recorded.

3.1.1.4. *Native perennial species richness site comparison*

Native perennial species richness was slightly higher at Acacia Woodland impact sites than control sites, with mean values of 35.50 and 33.50 species respectively. However, the Welch independent-samples t-test indicated that this difference was not statistically significant ($t = -0.20$, $df = 1.71$, $p = 0.864$). The 95% confidence interval for the difference in mean perennial species richness ranged from 53.38 to 49.38 species, indicating substantial uncertainty around the estimated difference, and given the very small sample size, these results should be interpreted cautiously.

Perennial species richness was significantly higher at Chenopod Shrubland control sites than at impact sites, with mean values of 35.20 and 27.29 species respectively. The Welch independent-samples t-test indicated a statistically significant difference between the two groups ($t = 2.45$, $df = 7.65$, $p = 0.041$). The 95% confidence interval for the difference in mean perennial species richness ranged from 0.41 to 15.42 species, indicating that control sites had, on average, between approximately 0.4 and 15.4 more perennial species than impact sites.

For Mallee Woodland, only impact site data was available in 2026, with a mean perennial species richness of 38 species. No statistical analysis was conducted as there was no control sites for comparison.

3.1.1.5. *Native perennial species richness comparison between years*

In Acacia Woodland impact sites native perennial species richness was relatively stable from 2017-2022, with consideration to seasonal variation (Figure 4). However, a decreasing trend in perennial species richness was observed from 2021-2023. From 2024-2026 perennial species richness in Acacia Woodland impact sites has been the greatest since the survey began in 2017, ranging from 30-34 species.

In Acacia Woodland control sites, a declining trend in perennial species richness was observed from 2017 to 2020, followed by an increasing trend from 2021-23 (Figure 4). Native perennial species richness in 2023 was the lowest recorded. From 2024-2026 native perennial species richness has been consistent, with averages between 33-35 species.

In Chenopod Shrubland impact sites a declining trend in native perennial species richness was observed from 2017 to 2020, followed by an increasing trend from 2021-22 (Figure 4). Native perennial species richness in 2023 was the lowest recorded. From 2024-2026, native perennial species richness in Chenopod Shrubland impact sites has decreased each year.

In Chenopod Shrubland control sites, perennial species richness had a declining trend from 2017 to 2020 (Figure 4). Perennial species richness in 2023 was the lowest recorded. From 2024-2026 native perennial species richness has been consistent, with averages between 30-36 species.

In Mallee Woodland impact sites, native perennial species richness was consistent from 2017-2019 (Figure 4). native perennial species richness increased by 13 species from 2019-2020. From 2020-2026 native perennial species richness in Mallee Woodland impact sites has been consistent, with averages between 33 and 45.

In Mallee Woodland control sites, native perennial species richness was consistent from 2017-2023 with averages between 17-21 species, with the exception of the 2021 survey which recorded 28 species (Figure 4). From 2024-2025 native perennial species richness in Mallee Woodland control sites was consistent.

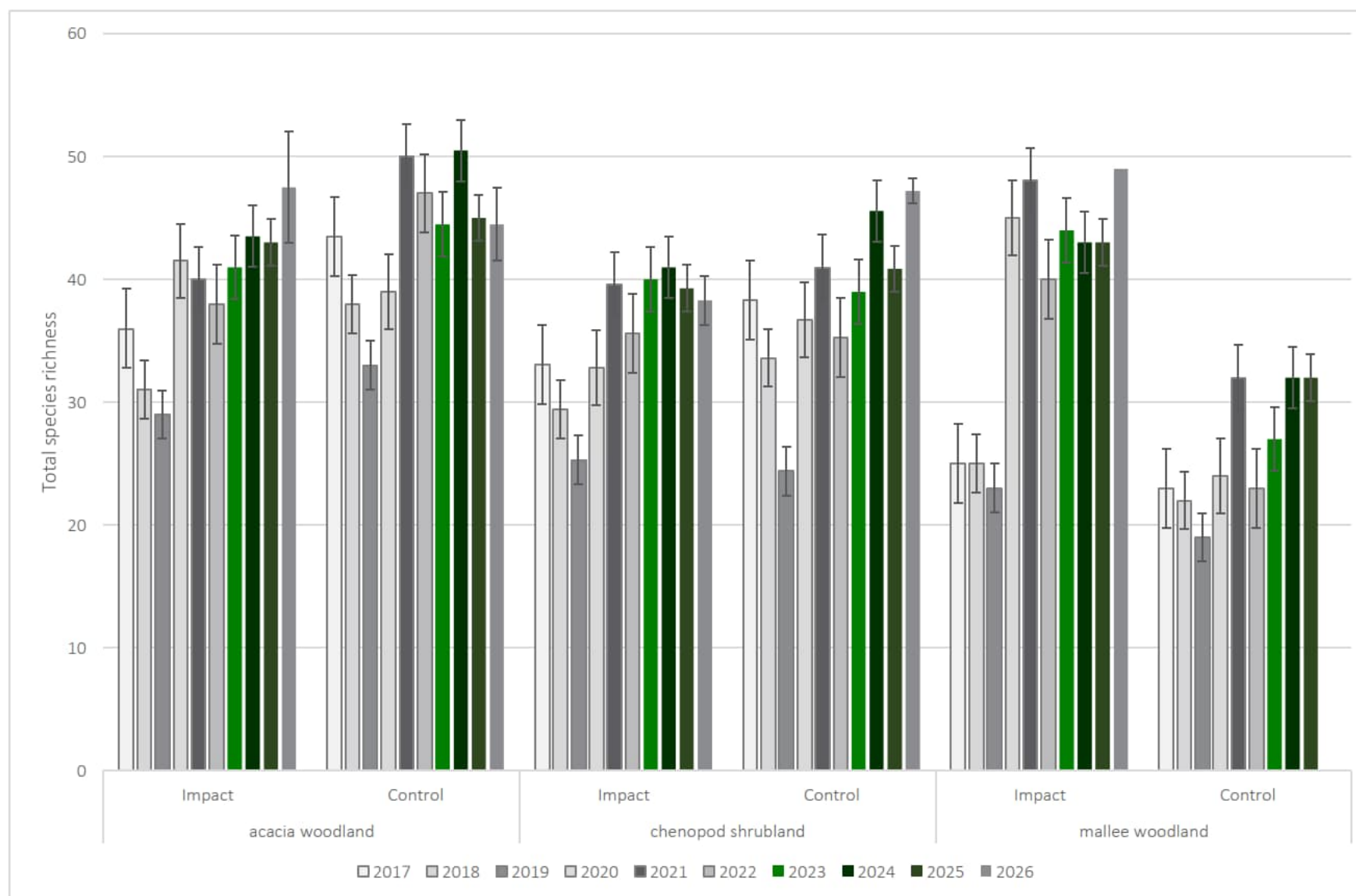


Figure 3: Total mean species richness (± SE) across the 2017-2026 monitoring period

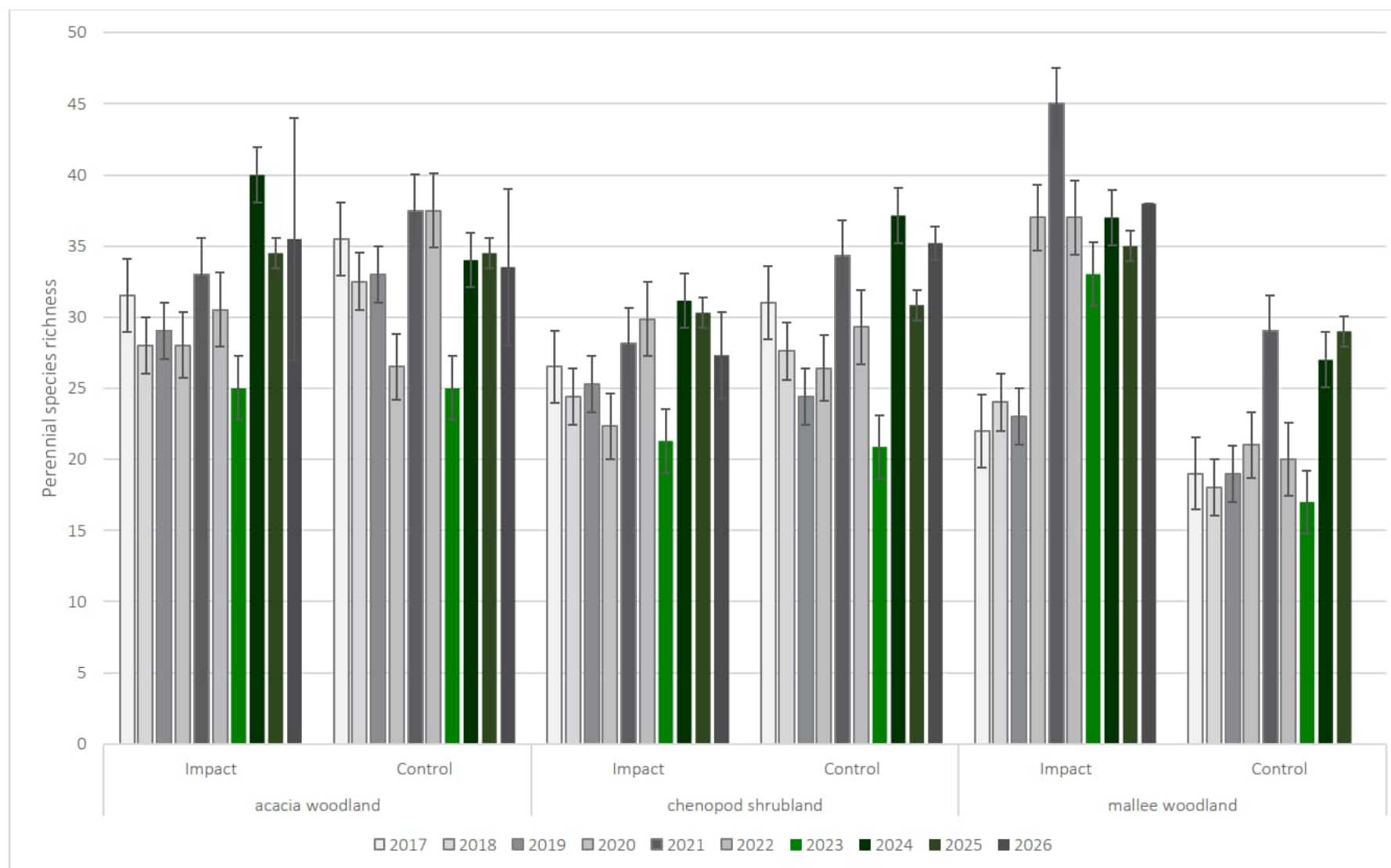


Figure 4: Mean perennial species diversity ($\pm$ SE) across the 2017-2026 monitoring period

3.1.2. Vegetation cover

3.1.2.1. Total species cover between sites

Mean all-species vegetation cover was higher at Acacia Woodland control sites than at impact sites (74.50% and 62.50%, respectively). However, the Welch independent-samples t-test indicated that this difference was not statistically significant ($t = 0.65$, $df = 1.90$, $p = 0.586$). The 95% confidence interval for the difference in mean cover ranged from -71.72 to 95.72 percentage points, indicating considerable uncertainty around the estimated difference. Given the very small sample size, these results should be interpreted cautiously.

Mean all-species vegetation cover was higher at Chenopod Shrubland control sites than at impact sites (60.21% and 45.63%, respectively). However, the Welch independent-samples t-test indicated that this difference was not statistically significant ($t = 1.57$, $df = 8.67$, $p = 0.152$). The 95% confidence interval for the difference in mean cover ranged from -6.54 to 35.69 percentage points, indicating uncertainty around the estimated difference.

In Mallee Woodland, only impact-site data were available in 2026, which had a total vegetation cover of 68%. No statistical analysis was conducted as there was no control data collected in 2026.

3.1.2.2. Total species cover between years

In Acacia Woodland impact sites (Figure 5), native species cover was relatively consistent from 2017 to 2021, with vegetation cover ranging between 28% and 40%. From 2022 to 2023, vegetation cover increased before decreasing during the 2024 and 2025 surveys. The 2026 survey recorded native vegetation cover of 62.5%, the highest recorded for Acacia Woodland impact sites. In Acacia Woodland control sites (Figure 5), native species cover showed a decreasing trend from 2017 to 2019. From 2020 to 2021, vegetation cover was relatively consistent before decreasing in 2022. The 2023 survey recorded the highest mean vegetation cover since the survey began in 2017. Similar to the impact sites, Acacia Woodland control sites had relatively lower species cover during the 2024 and 2025 surveys. The 2026 survey recorded native vegetation cover of 74.5% within Acacia Woodland control sites, representing an increase of approximately 27% from the 2025 survey.

In Chenopod Shrubland impact sites (Figure 5), native species cover was variable across the survey period. A decreasing trend was observed from 2017 to 2019. This was followed by an increase in the 2020 survey, and a decrease in 2021. Native species cover increased in both 2022 and 2023 before declining again in 2024 and 2025. In 2026, native species cover in Chenopod Shrubland control sites (Figure 5) was 45.63%, which was the highest recorded since surveying began. A similar pattern was observed in Chenopod Shrubland control sites, although native species cover generally had higher averages at control sites than at impact sites.

In Mallee Woodland impact sites (Figure 5), native species cover was consistent from 2017 to 2019. An increase in native species cover was observed in 2020, followed by a decreasing trend from 2020 to 2022. Native species cover increased in the 2023 survey, followed by a decrease in 2024 and 2025. In 2026, native species cover within Mallee Woodland impact sites was recorded at 68%, which was the highest recorded since surveying began in 2017. Within Mallee Woodland control sites (Figure 5), a decreasing trend in native species cover was observed from the 2017 to 2021 surveys before stabilising in 2022 and increasing in the 2023 survey. In 2024, Mallee Woodland control sites recorded their lowest native species cover, at 2.4%, before increasing in 2025. No data were collected during the 2026 survey.

3.1.2.3. *Native perennial species cover between sites*

Mean perennial vegetation cover was higher at Acacia Woodland control sites than at impact sites (73.22% and 59.06%, respectively). However, the Welch independent-samples t-test indicated that this difference was not statistically significant ($t = 0.78$, $df = 1.84$, $p = 0.521$). The 95% confidence interval for the difference in mean cover ranged from -70.17 to 98.49 percentage points, indicating substantial uncertainty around the estimated difference. Given the very small sample size, these results should be interpreted cautiously.

Mean perennial vegetation cover was higher at Chenopod Shrubland control sites than at impact sites (58.85% and 41.37%, respectively). However, the Welch independent-samples t-test indicated that this difference was not statistically significant ($t = 1.98$, $df = 8.79$, $p = 0.080$). The 95% confidence interval for the difference in mean cover ranged from -2.56 to 37.52 percentage points, indicating uncertainty around the estimated difference.

In Mallee Woodland, the single impact site had a perennial cover of 54.2%. No statistical analysis was conducted as there was no control data collected in 2026.

3.1.2.4. *Native perennial species cover between years*

In Acacia Woodland impact sites (Figure 5), perennial species cover decreased from 34.9% in 2017 to 29.4% in 2019. This was followed by an increasing trend, reaching 44.84% in 2022, before declining through 2023 and 2024. Perennial species cover remained relatively stable between 2024 and 2025, at 33.15% and 34.70%, respectively. In 2026, perennial species cover increased substantially to 59.1%, representing the highest value recorded for Acacia Woodland impact sites during the survey period. Acacia Woodland control sites (Figure 5) showed a decreasing trend from 70.3% in 2017 to 58.8% in 2019, followed by an increase to 76.95% in 2021. Perennial species cover subsequently declined to 44.11% in 2023 and remained relatively stable in 2024 and 2025. In 2026, perennial species cover increased to 73.2%, although this remained slightly below the 2021 value. Perennial species cover was consistently higher at control sites than at impact sites throughout the survey period.

In Chenopod Shrubland impact sites, perennial species cover decreased from 21.7% in 2017 to 16.7% in 2018 before increasing slightly in 2019. Perennial species cover increased in 2020, declined in 2021, and then reached 32.49% in 2022. This was followed by a decreasing trend from 2023 to 2025, when perennial species cover was recorded at 21.32%. In 2026, perennial species cover increased substantially to 41.4%, the highest value recorded for Chenopod Shrubland impact sites since surveying began. Chenopod Shrubland control sites (Figure 5) showed a decrease in perennial species cover from 31.8% in 2017 to 27.2% in 2019, followed by an increasing trend from 2020 to 2022. Perennial species cover then declined to 22.84% in 2023, increased in 2024, and remained relatively stable in 2025. In 2026, perennial species cover increased to 58.8%, the highest value recorded for Chenopod Shrubland control sites. Perennial species cover was generally higher at control sites than at impact sites.

In Mallee Woodland impact sites (Figure 5), perennial species cover remained relatively stable between 2017 and 2019 before increasing to 16.7% in 2020. This was followed by a decline from 2021 to 2022, when perennial species cover reached a low of 6.42%. Perennial species cover increased to 14.19% in 2023, declined again in 2024, and then increased slightly in 2025. In 2026, perennial species cover increased substantially to 54.2%, the highest value recorded for Mallee Woodland impact sites during the survey period. Mallee Woodland control sites (Figure 5) showed a general decreasing trend from 16.15% in 2017 to 8.64% in 2021, with minor fluctuations between 2022 and 2023. Perennial species cover declined to its lowest recorded value of 2.06% in 2024 before increasing to 8.2% in 2025.

In 2026, perennial species cover was recorded at 0%, representing a further decline from the previous survey. Consequently, perennial species cover was substantially higher at Mallee Woodland impact sites than at control sites in 2026.

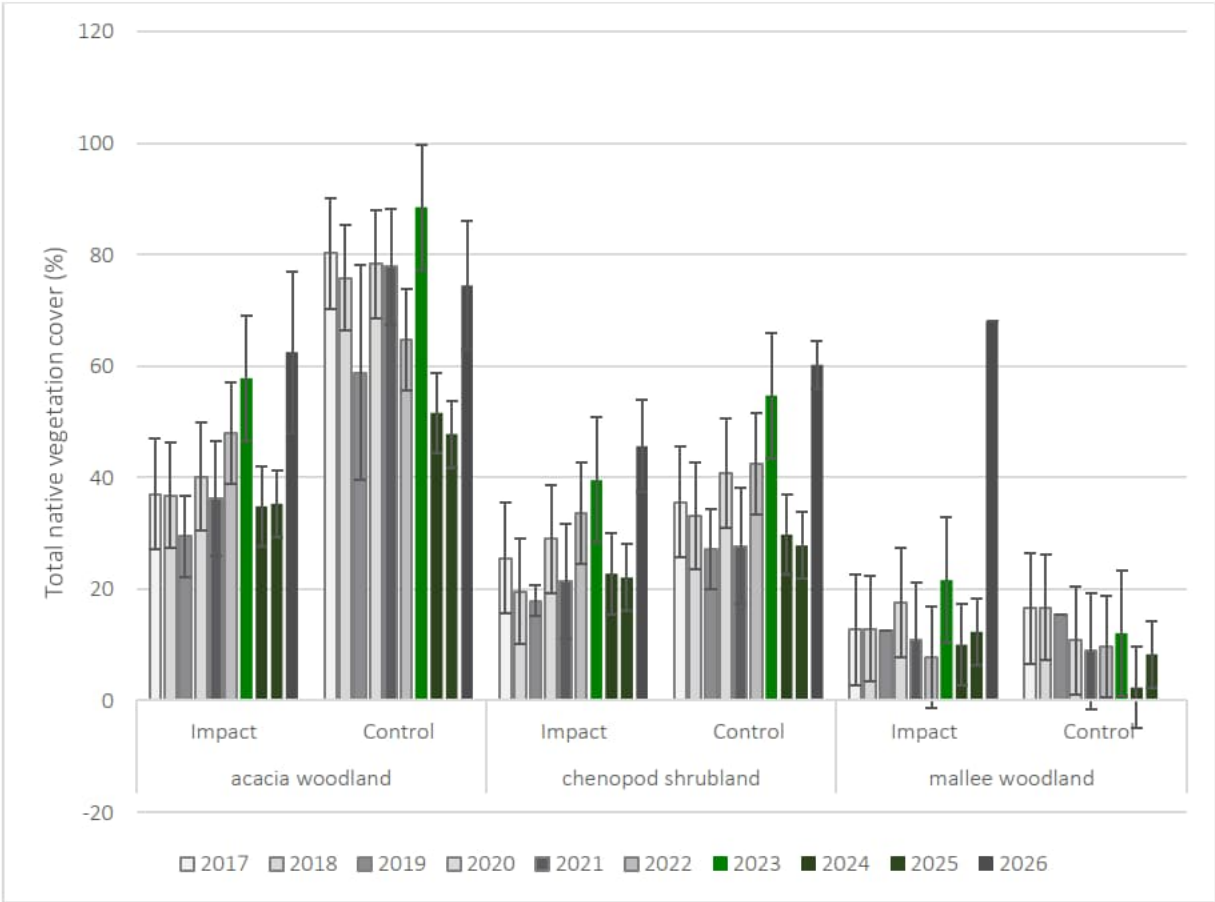


Figure 5: Mean total vegetation cover ($\pm$ SE) across the 2017-2026 monitoring period

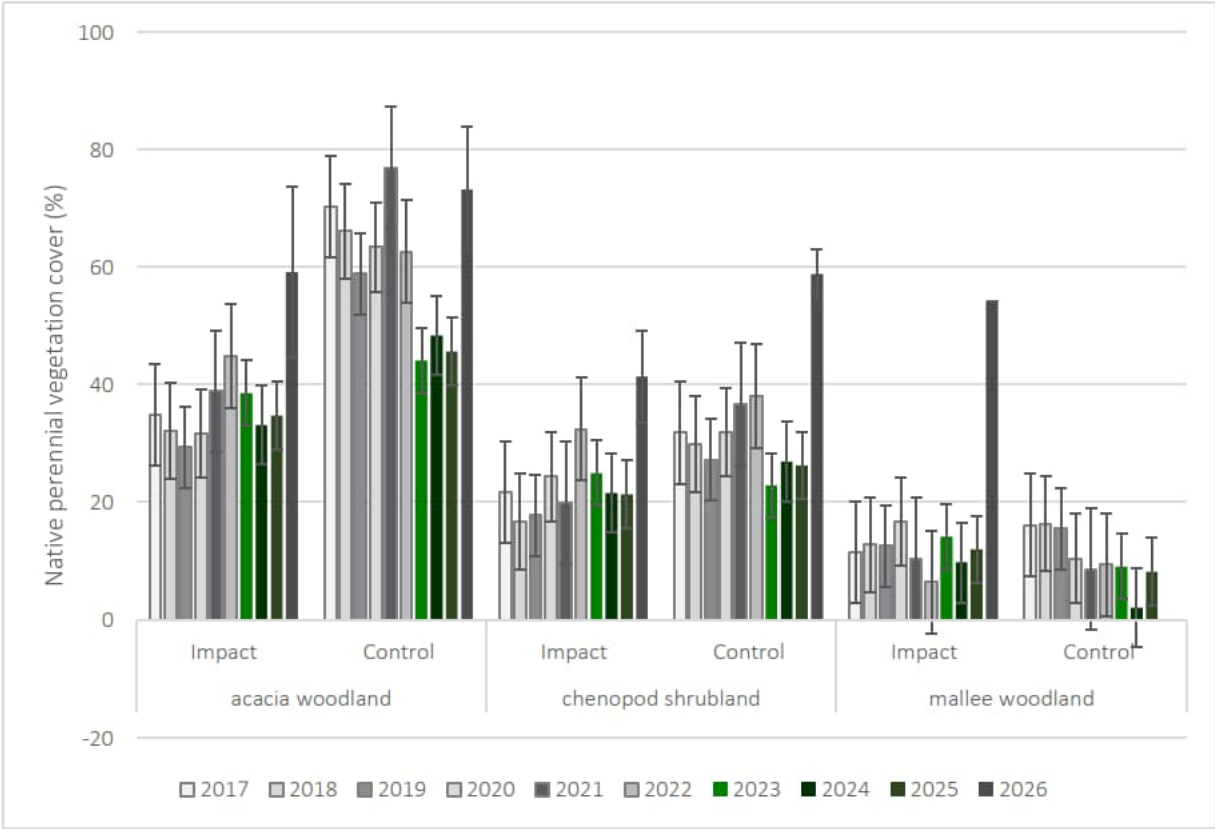


Figure 6: Mean perennial vegetation cover ($\pm$ SE) across the 2017-2026 monitoring period

3.1.3. Perennial density

3.1.3.1. Native perennial density between sites

When observing patterns in perennial density between Jessup transects in control versus impact sites, the 2026 data, show that impact sites recorded a lower perennial plant density than control sites in both Acacia Woodland and Chenopod Shrubland.

Mean total perennial stem density was higher at Acacia Woodland control sites than at impact sites, with mean densities of 0.331 and 0.183, respectively. However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = 0.85$, $df = 1.29$, $p = 0.525$). The 95% confidence interval for the difference in mean stem density ranged from -1.19 to 1.49.

Mean mature perennial density was also higher at Acacia Woodland control sites than at impact sites, with mean densities of 0.293 and 0.160, respectively. The Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = 0.77$, $df = 1.16$, $p = 0.566$). The 95% confidence interval for the difference in mean mature perennial density ranged from -1.44 to 1.70.

Mean juvenile perennial density was higher at Acacia Woodland control sites than at impact sites, with mean densities of 0.088 and 0.056, respectively. However, this difference was not statistically significant according to the Welch independent-samples *t*-test ($t = 0.52$, $df = 1.21$, $p = 0.684$). The 95% confidence interval for the difference in mean juvenile perennial density ranged from -0.48 to 0.55. Given the very small sample sizes, all three results should be interpreted cautiously.

Mean total perennial stem density was higher at Chenopod Shrubland control sites than at impact sites, with mean densities of 0.349 and 0.283, respectively. However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = 0.79$, $df = 6.20$, $p = 0.461$). The 95% confidence interval for the difference in mean stem density ranged from 0.137 to 0.269.

Mean mature perennial density was higher at Chenopod Shrubland control sites than at impact sites, with mean densities of 0.302 and 0.227, respectively. However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = 1.30$, $df = 7.02$, $p = 0.235$). The 95% confidence interval for the difference in mean mature perennial density ranged from -0.062 to 0.212.

Mean juvenile perennial density was slightly higher at Chenopod Shrubland control sites than at impact sites, with mean densities of 0.168 and 0.144, respectively. This difference was not statistically significant according to the Welch independent-samples *t*-test ($t = 0.53$, $df = 8.11$, $p = 0.608$). The 95% confidence interval for the difference in mean juvenile perennial density ranged from -0.078 to 0.125.

3.1.3.2. Native perennial density between years

Perennial density varied between vegetation types and survey years (Figure 7). The clearest changes were observed in mature perennial density within Acacia Woodland and Chenopod Shrubland, and in juvenile perennial density at Acacia Woodland impact sites in 2026.

Acacia woodland

Mature perennial density at Acacia Woodland impact sites increased from 0.121 plants/m² in 2020 to 0.253 plants/m² in 2022, before declining to 0.111 plants/m² in 2023. Density then increased to a survey-period maximum of 0.360 plants/m² in 2024, followed by a decline to 0.161 plants/m² in 2025 and 0.160 plants/m² in 2026. At control sites, mature perennial density declined slightly from 0.251 plants/m² in 2020 to 0.221 plants/m² in 2021, before increasing to 0.488 plants/m² in 2022 and 0.629 plants/m² in 2023. This was followed by a decrease in 2024, with density remaining relatively stable in

2025 and 2026 at 0.308 and 0.293 plants/m², respectively. Mature perennial density was generally higher at control sites than at impact sites, except in 2024.

Juvenile perennial density at Acacia Woodland impact sites remained relatively low and stable between 2020 and 2025, ranging from 0.050 to 0.085 plants/m². In 2026, juvenile density increased substantially to 0.563 plants/m², the highest value recorded during the survey period. At control sites, juvenile density declined from 0.215 plants/m² in 2020 to 0.064 plants/m² in 2021, before increasing to 0.389 plants/m² in 2022. Density subsequently declined in 2023 and 2024 before increasing slightly in 2025 and 2026. Juvenile density was higher at control sites than impact sites during most survey years, although impact sites recorded a substantially higher value in 2026.

Chenopod Shrubland

Mature perennial density at Chenopod Shrubland impact sites increased from 0.130 plants/m² in 2020 to 0.378 plants/m² in 2022, before declining to 0.201 plants/m² in 2023. Density increased again to a survey-period maximum of 0.402 plants/m² in 2024, followed by a decrease to 0.188 plants/m² in 2025. In 2026, mature perennial density increased slightly to 0.227 plants/m². At control sites, mature perennial density declined substantially from 0.241 plants/m² in 2020 to 0.032 plants/m² in 2021 and 0.048 plants/m² in 2022. Density then increased sharply to 0.321 plants/m² in 2023, declined to 0.244 plants/m² in 2024, and increased again in 2025 and 2026 to 0.298 and 0.302 plants/m², respectively. Mature perennial density was higher at impact sites in 2021, 2022 and 2024, while control sites recorded higher values in 2020, 2023, 2025 and 2026.

Juvenile perennial density at Chenopod Shrubland impact sites remained consistently low between 2020 and 2023, ranging from 0.025 to 0.045 plants/m². Density increased substantially to 0.203 plants/m² in 2024, before declining to 0.025 plants/m² in 2025. In 2026, juvenile density increased again to 0.144 plants/m². At control sites, juvenile density fluctuated between 2020 and 2022, ranging from 0.060 to 0.091 plants/m², before declining to 0.030 plants/m² in 2023 and 0.024 plants/m² in 2024. Density increased in 2025 and reached a survey-period maximum of 0.168 plants/m² in 2026. Control sites generally recorded higher juvenile density than impact sites, except in 2024, when impact sites had substantially greater density.

Mallee Woodland

Mature perennial density at Mallee Woodland impact sites fluctuated across the survey period. Density declined from 0.260 plants/m² in 2020 to 0.215 plants/m² in 2021, before increasing to a maximum of 0.338 plants/m² in 2022. Density then declined to 0.228 plants/m² in 2023 and decreased sharply to 0.018 plants/m² in 2024. This was followed by a substantial recovery to 0.195 plants/m² in 2025 and 0.258 plants/m² in 2026. At control sites, mature perennial density remained very low between 2020 and 2023, ranging from 0.008 to 0.013 plants/m². Density increased sharply to 0.195 plants/m² in 2024 before declining to 0.018 plants/m² in 2025 and 0 plants/m² in 2026. Mature perennial density was higher at impact sites during all years except 2024.

Juvenile perennial density at Mallee Woodland impact sites decreased from 0.105 plants/m² in 2020 to 0.063 plants/m² in 2021, followed by a slight increase to 0.090 plants/m² in 2022. Density then declined to 0.033 plants/m² in 2023 and 0.005 plants/m² in 2024, remaining unchanged in 2025. In 2026, juvenile density increased slightly to 0.033 plants/m². At control sites, juvenile perennial density was 0 plants/m² from 2020 to 2022, increased to 0.015 plants/m² in 2023, and then declined to 0.005 plants/m² in 2024 and 2025 before returning to 0 plants/m² in 2026. Juvenile perennial density was consistently higher at impact sites than at control sites throughout the survey period.

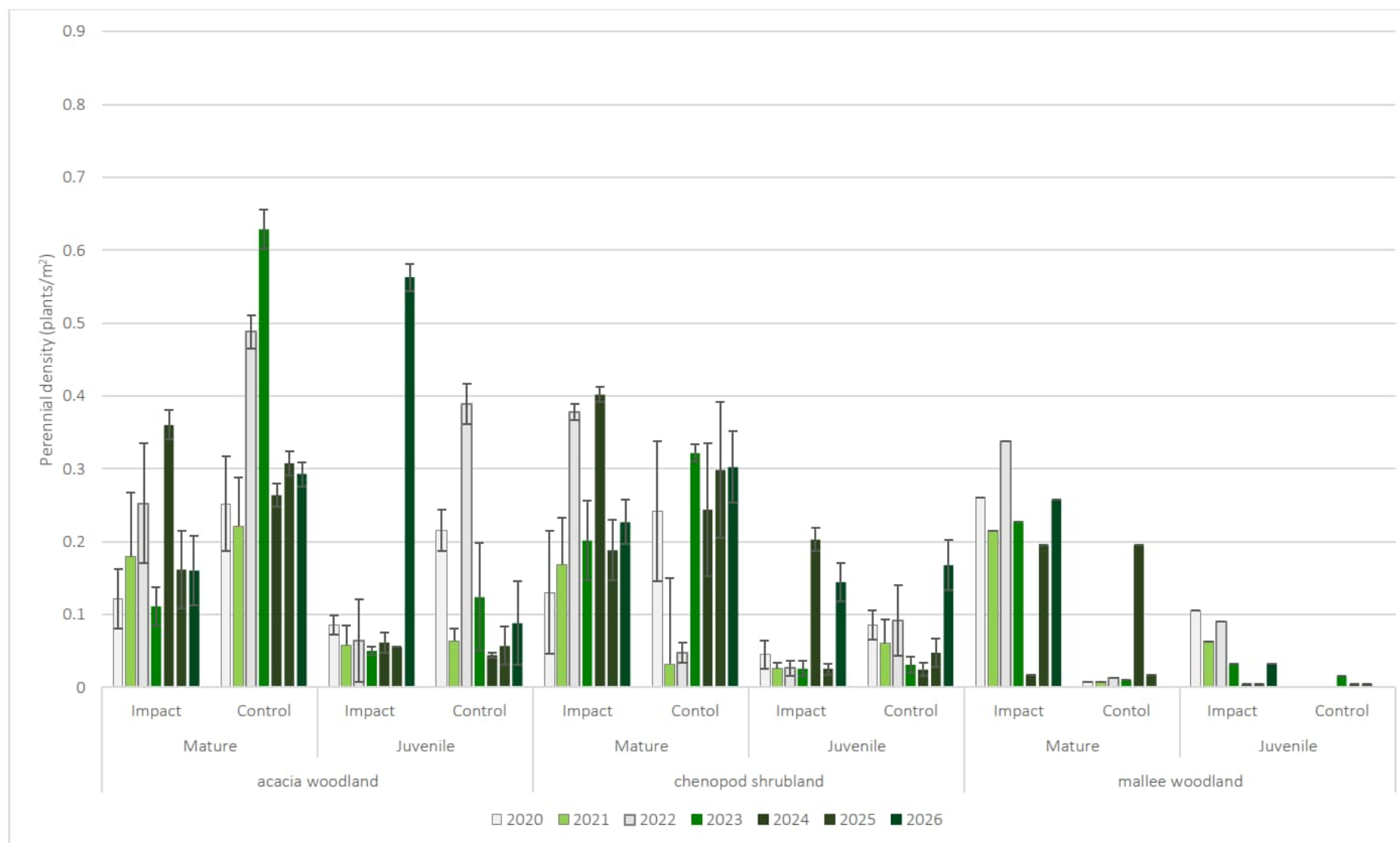


Figure 7: Mean density ($\pm$ SE) of adult and juvenile plants based on Jessup transect across the 2020-2026 monitoring period

3.1.4. Native recruitment

3.1.4.1. Native recruitment between sites

The mean proportion of quadrats containing recruits was higher at Acacia Woodland impact sites than at control sites (66.0% and 50.5%, respectively). However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = -1.72$, $df = 1.74$, $p = 0.246$). The 95% confidence interval for the difference in mean recruitment ranged from -60.34 to 29.34 percentage points, indicating considerable uncertainty around the estimated difference. Given the very small sample size, these results should be interpreted cautiously.

The mean proportion of quadrats containing recruits was slightly higher at Chenopod Shrubland impact sites than at control sites (64.0% and 60.2%, respectively). However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = -0.47$, $df = 9.49$, $p = 0.648$). The 95% confidence interval for the difference in mean recruitment ranged from -21.89 to 14.29 percentage points, indicating uncertainty around the estimated difference.

3.1.4.2. Native recruitment between years

The values presented represent the percentage of native perennial species that were recruiting during each survey year. Native perennial recruitment varied considerably among vegetation types and between impact and control sites (Figure 8). Across most vegetation types, recruitment was relatively high in 2017, declined substantially in 2018 and 2019, and increased again in 2020. Recruitment continued to fluctuate between 2021 and 2026, with the relative difference between impact and control sites varying among vegetation types.

In Acacia Woodland impact sites, the percentage of native perennial species recruiting declined from 90.81% in 2017 to 18.70% in 2019. Recruitment then increased to 68.50% in 2020 and remained relatively high through 2022, before declining to 54.98% in 2023. Recruitment increased to 72.00% in 2024, declined to 58.00% in 2025 and increased again to 66.00% in 2026. At Acacia Woodland control sites, recruitment declined from 88.48% in 2017 to 13.52% in 2019, followed by an increase to 68.00% in 2020. Recruitment remained relatively high between 2021 and 2024, ranging from 59.50% to 67.00%, before declining to 41.00% in 2025 and increasing to 50.50% in 2026. The percentage of recruiting native perennial species was higher at impact sites than control sites in 2017, 2019, 2020, 2022 and 2024, while control sites recorded higher values in 2018, 2021, 2023, 2025 and 2026.

In Chenopod Shrubland impact sites, native perennial recruitment declined substantially from 88.40% in 2017 to 23.00% in 2018 and 17.41% in 2019. Recruitment increased to 71.10% in 2020 before declining to 56.46% in 2021. Recruitment then increased to 61.86% in 2022 and 66.84% in 2023, followed by a gradual decline to 65.43% in 2024, 61.00% in 2025 and 60.20% in 2026. At Chenopod Shrubland control sites, recruitment declined from 86.84% in 2017 to 11.49% in 2019 before increasing to 69.40% in 2020. Recruitment remained relatively high between 2021 and 2023, reaching 70.29% in 2022, before declining to 60.86% in 2024. Recruitment then increased slightly to 62.57% in 2025 and 64.00% in 2026. Control sites generally had a higher percentage of recruiting native perennial species than impact sites from 2021 to 2023 and from 2025 to 2026, while impact sites recorded higher recruitment in 2020 and 2024.

In Mallee Woodland impact sites, the percentage of native perennial species recruiting decreased from 80.00% in 2017 to 28.00% in 2018 and 17.36% in 2019. Recruitment increased to 64.00% in 2020 before

declining to 48.00% in 2021 and 35.00% in 2022. Recruitment subsequently increased to 45.45% in 2023 and 63.00% in 2024, before remaining relatively stable at 61.00% in both 2025 and 2026. At Mallee Woodland control sites, recruitment declined from 91.30% in 2017 to 15.79% in 2019, followed by an increase to 63.00% in 2020 and 68.75% in 2021. Recruitment then declined to 52.00% in 2022, increased to 77.78% in 2023 and declined to 59.00% in 2024 and 38.00% in 2025. No native perennial species were recorded as recruiting at Mallee Woodland control sites in 2026. The percentage of recruiting native perennial species was higher at impact sites than control sites in 2024, 2025 and 2026.

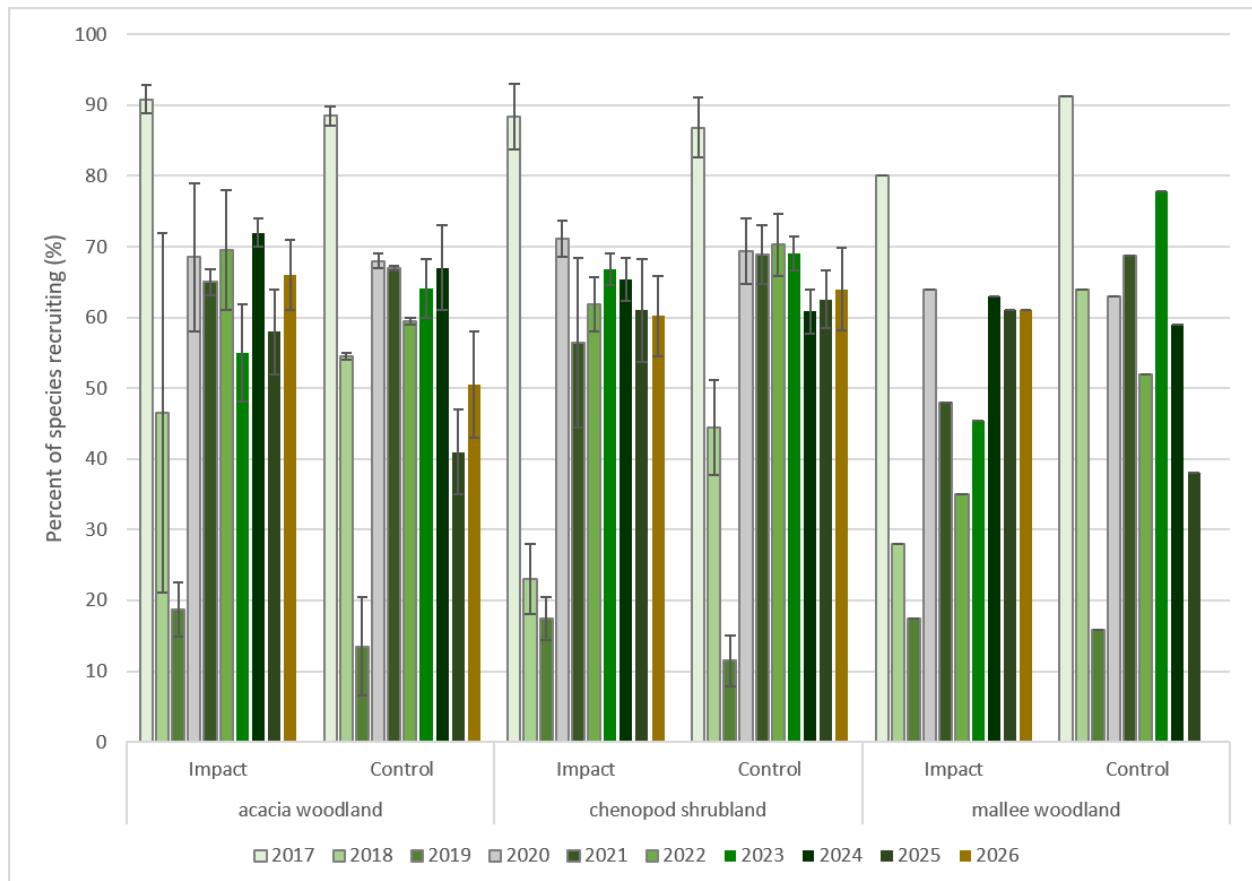


Figure 8: Mean percentage ($\pm$ SE) of species recruiting across the 2017-2026 monitoring period

3.1.5. Subplot monitoring (cover estimates and perennial recruitment)

3.1.5.1. Acacia woodland

In 2026, the mean **grass cover** was higher at Acacia Woodland control sites (32.5%) than at impact sites (12.5%). However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = 1.57$, $df = 1.08$, $p = 0.348$). The 95% confidence interval for the difference in mean grass cover ranged from -116.23 to 156.23 percentage points, indicating uncertainty around the estimated difference. Overall, grass cover was generally higher at control sites than at impact sites (Figure 9). Grass cover at impact sites declined from 12.6% in 2017 to a minimum of 1.8% in 2019. It then increased to 18.5% in 2022 before fluctuating between 9.0% and 18.5% from 2023 to 2025. In 2026, grass cover declined to 12.5%. At control sites, grass cover decreased from 22.7% in 2017 to 0.6% in 2019, followed by an increase to 40.0% in 2022. Grass cover then declined to 19.0% in 2023 before increasing to 32.5% in 2026, the highest value recorded since 2022.

In 2026, the mean cover of **bare ground** was slightly higher at Acacia Woodland impact sites (37.5%) than at control sites (32.5%). However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = -0.21$, $df = 1.22$, $p = 0.863$). The 95% confidence interval for the difference in mean bare-ground cover ranged from -204.08 to 194.08 percentage points, indicating uncertainty around the estimated difference. Since survey began, the overall cover of bare ground in Acacia Woodland has been variable between survey seasons at both impact and control sites, with no overall trends within the data (Figure 9). Within impact sites, it increased from 49.4% in 2017 to a maximum of 70.0% in 2020. It decreased to 52.5% in 2022 before increasing to 63.5% in 2024. Bare-ground cover then decreased to 52.5% in 2025 and 37.5% in 2026, the lowest value recorded at impact sites since 2017. At control sites, bare-ground cover declined from 39.8% in 2017 to 27.1% in 2018, increased to 70.1% in 2019, and then declined to 22.5% in 2024. It remained relatively low in 2025 before increasing to 32.5% in 2026. Bare-ground cover was generally higher at impact sites than at control sites, with the exception of the 2019 survey.

Mean **litter cover** was higher at Acacia Woodland control sites (7.50%) than at impact sites (5.25%). However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = 0.42$, $df = 1.51$, $p = 0.727$). The 95% confidence interval for the difference in mean litter cover ranged from -29.62 to 34.12 percentage points, indicating uncertainty around the estimated difference. Overall, litter cover was generally higher at impact sites, although control sites recorded higher values in 2020, 2021 and 2026 (Figure 9). Litter cover at impact sites declined from 29.5% in 2017 to 14.4% in 2020 and decreased further to 2.5% in 2021. Litter cover increased to 14.0% in 2022 and remained relatively stable through 2024 before declining to 12.8% in 2025 and 5.3% in 2026. At control sites, litter cover fluctuated between 2017 and 2020, increasing from 10.7% in 2017 to 20.5% in 2020. It then declined to 3.0% in 2022 before increasing to 9.5% in 2023. Litter cover declined to 5.0% in 2024 and increased slightly to 7.5% in 2026.

Scalded surface cover was recorded as 0% at both Acacia Woodland control and impact sites. As there was no variation in the data, the Welch independent-samples *t*-test could not be calculated. Overall, scalded-surface cover was absent or occurred at very low levels throughout the survey period (Figure 9). Scalded-surface cover was recorded as 0% at both impact and control sites from 2017 to 2021. Low levels were recorded in 2022, at 1.0% for impact sites and 0.5% for control sites. In 2023, scalded-surface cover was 0% at impact sites but increased to 4.0% at control sites. In 2024, scalded-surface cover was recorded at 0.25% at impact sites and 0% at control sites. No scalded-surface cover was recorded at either impact or control sites in 2025 or 2026.

Soil-crust cover was recorded as 0% at both Acacia Woodland control and impact sites (Figure 9). As there was no variation in the data, the Welch independent-samples *t*-test could not be calculated. Soil-crust cover remained low and variable at impact sites, ranging from 0% to 7.5%. It increased from 1.4% in 2017 to 2.2% in 2018, declined to 0% in 2019 and 2020, and increased to 7.5% in 2022. Soil-crust cover was recorded as 0% from 2023 to 2026. At control sites, soil-crust cover was highest in 2018, at 10.2%, before declining to 0% in 2019 and 2020. It increased slightly in 2021 and 2022, declined to 0% in 2025, and increased to 5.0% in 2026. Soil-crust cover was generally higher at control sites than at impact sites, except in 2022.

Mean **perennial recruitment** was higher at Acacia Woodland control sites than at impact sites, with mean values of 2.0 and 1.00 recruits, respectively. However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = 1.00$, $df = 1.00$, $p = 0.500$). The 95% confidence interval for the difference in mean recruitment ranged from 11.71 to 13.71 recruits. Given the very small sample size, this result should be interpreted cautiously.

3.1.5.2. *Chenopod shrubland*

Mean **bare-ground cover** was higher at Chenopod Shrubland impact sites (65.0%) than at control sites (52.0%). However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = -0.84$, $df = 6.53$, $p = 0.431$). The 95% confidence interval for the difference in mean bare-ground cover ranged from -50.16 to 24.16 percentage points. Since the survey began in 2017, bare-ground cover was generally higher at impact sites than at control sites, except in 2019, 2021 and 2022 (Figure 10Figure 9). Bare-ground cover at impact sites increased from 70.5% in 2017 to 83.5% in 2018 and 84.1% in 2019. It then declined to 76.0% in 2020 and 55.7% in 2021. Bare-ground cover increased to 61.7% in 2022 and 76.9% in 2023 before gradually declining to 74.2% in 2024, 72.0% in 2025 and 65.0% in 2026. At control sites, bare-ground cover increased from 65.1% in 2017 to 88.9% in 2019, the highest value recorded for control sites. It subsequently declined to 72.8% in 2020 and increased slightly to 65.7% in 2022. Bare-ground cover then declined to 64.2% in 2024 and 52.0% in 2026, the lowest value recorded during the survey period.

Mean **grass cover** was higher at Chenopod Shrubland impact sites (19.0%) than at control sites (12.6%). This difference was not statistically significant according to the Welch independent-samples *t*-test ($t = -0.80$, $df = 9.94$, $p = 0.444$). The 95% confidence interval for the difference in mean grass cover ranged from -24.30 to 11.50 percentage points. Over the survey period, grass cover at Chenopod Shrubland impact sites decreased from 9.8% in 2017 to a minimum of 0.5% in 2019 (Figure 10Figure 9). It then increased to 6.2% in 2020 and 14.7% in 2021, reaching a survey-period maximum of 19.1% in 2022. Grass cover subsequently declined to 7.4% in 2023, increased to 13.0% in 2024, and declined again to 8.0% in 2025. In 2026, grass cover increased to 19.0%, equal to the highest value recorded for impact sites. At control sites, grass cover declined from 8.8% in 2017 to 3.8% in 2018 and 0% in 2019, before increasing to 7.6% in 2020. Grass cover then increased to 16.6% in 2022 and reached a maximum of 20.5% in 2024. This was followed by a decline to 10.7% in 2025 and a slight increase to 12.6% in 2026. Grass cover was higher at impact sites than at control sites in 2017, 2018, 2021, 2022 and 2026 (Figure 10Figure 9).

Mean **litter cover** was higher at Chenopod Shrubland control sites than at impact sites (11.40% and 1.57%, respectively). However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = 1.01$, $df = 4.03$, $p = 0.368$). The 95% confidence interval for the difference in mean litter cover ranged from -17.04 to 36.70 percentage points. Litter cover at impact sites fluctuated throughout the survey period (Figure 10Figure 9), declining from 6.8% in 2017 to 5.7% in 2019 before increasing to 8.2% in 2020Figure 9. Litter cover then decreased to 2.6% in 2021, increased to 5.6% in 2022 and declined to 2.8% in 2024. It increased to 7.4% in 2025 before declining to 1.6% in 2026, the lowest value recorded for impact sites. At control sites, litter cover was relatively high in 2017 and 2018, at 10.7% and 11.1%, respectively, before declining to 4.2% in 2019. Litter cover increased to 6.6% in 2020, declined to 3.6% in 2023 and 3.5% in 2024, and then increased to 10.8% in 2025 and 11.4% in 2026. Litter cover was generally higher at control sites than at impact sites, particularly in 2026.

Mean **soil-crust cover** was higher at Chenopod Shrubland control sites than at impact sites (4.8% and 1.6%, respectively). However, this difference was not statistically significant according to the Welch independent-samples *t*-test ($t = 1.16$, $df = 4.52$, $p = 0.303$). The 95% confidence interval for the difference in mean soil-crust cover ranged from -4.14 to 10.60 percentage points. Soil-crust cover has remained low at both impact and control sites throughout the survey period (Figure 10Figure 9). At impact sites, soil-crust cover was absent in 2017, 2019, 2020 and 2023. It increased slightly to 0.21% in 2018, 0.71% in 2021 and 0.93% in 2022, reaching 1.67% in 2024. Soil-crust cover declined to 0.86% in 2025 before increasing to 1.57% in 2026. At control sites, soil-crust cover was highest in 2017, at 1.94%, before declining to 0.54% in 2018 and 0% in 2019 and 2020. It remained low between 2021 and 2025

before increasing substantially to 4.80% in 2026, the highest value recorded for either treatment. Soil-crust cover was higher at control sites in 2017, 2018, 2022, 2023, 2025 and 2026.

Mean **scalded-surface cover** was higher at Chenopod Shrubland impact sites than at control sites (0.14% and 0%, respectively). However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = -1.00$, $df = 6.00$, $p = 0.356$). The 95% confidence interval for the difference in mean scalded-surface cover ranged from -0.49 to 0.21 percentage points. Scalded-surface cover was absent at both impact and control sites from 2017 to 2021 (Figure 10Figure 9). At impact sites, scalded-surface cover increased to 0.71% in 2022, declined to 0% in 2023, increased slightly to 0.17% in 2024 and 0.43% in 2025, and returned to 0% in 2026. At control sites, scalded-surface cover increased to 1.00% in 2022, declined slightly to 0.14% in 2023 and then reached a survey-period maximum of 2.85% in 2024. It declined to 0.43% in 2025 and was absent in 2026. Overall, scalded-surface cover was absent or occurred at very low levels throughout the survey period, with no scalded surface recorded at either impact or control sites in 2026.

Mean **perennial recruitment** was higher at Chenopod Shrubland impact sites than at control sites, with mean values of 1.86 and 0.40 recruits, respectively. However, the Welch independent-samples *t*-test indicated that this difference was not statistically significant ($t = -2.04$, $df = 7.51$, $p = 0.078$). The 95% confidence interval for the difference in mean recruitment ranged from -3.12 to 0.21 recruits, indicating uncertainty around the estimated difference. Although the result suggested a potential difference between impact and control sites, it did not meet the significance threshold of $p \leq 0.05$.

3.1.5.3. *Mallee Woodland*

In 2026, Mallee Woodland was represented by one impact site only (PH15), with no corresponding control site available for comparison. At this site, grass cover was 10%, bare-ground cover was 40%, litter cover was 5%, and surface-crust cover was 5%. Scalded-surface cover and perennial recruitment were both recorded as 0%. As only one Mallee Woodland impact site was surveyed, independent-samples *t*-tests could not be conducted and no statistical comparison between impact and control sites was possible. These values are therefore reported descriptively only.

Grass cover at Mallee Woodland impact sites declined from 12.75% in 2017 to 0% in 2019 (Figure 10Figure 9). Grass cover then increased to 7.20% in 2020 before declining to 2.00% in 2021. A substantial increase occurred in 2022, when grass cover reached 45.00%, followed by a decline to 21.00% in 2023. Grass cover then remained relatively stable at 40.00% in both 2024 and 2025 before declining to 10.00% in 2026. At control sites, grass cover was 0% from 2017 to 2019, increased to 5.00% in 2020 and returned to 0% from 2021 to 2026. Grass cover was therefore consistently higher at impact sites than at control sites throughout the survey period.

Bare-ground cover at impact sites fluctuated between survey years (Figure 10Figure 9). It declined from 54.65% in 2017 to 47.65% in 2018 before increasing substantially to 95.75% in 2019, the highest value recorded for impact sites. Bare-ground cover then declined to 57.65% in 2020, 50.00% in 2021 and 27.00% in 2022. It increased to 55.00% in 2023 before declining to 43.00% in both 2024 and 2025 and 40.00% in 2026. At control sites, bare-ground cover increased from 71.20% in 2017 to 82.00% in 2022, before declining to 72.00% in 2024 and 2025. Bare-ground cover was recorded as 0% at control sites in 2026. Control sites generally had higher bare-ground cover than impact sites, except in 2019 and 2026.

Litter cover at impact sites declined from 16.50% in 2017 to 2.95% in 2019, before increasing substantially to 27.80% in 2020 (Figure 10Figure 9). Litter cover then declined to 1.00% in 2021 and increased to 6.00% in 2022. It subsequently decreased to 4.00% in 2023 and 2.00% in both 2024 and 2025, before increasing to 5.00% in 2026. At control sites, litter cover declined from 31.30% in 2017 to 17.80% in 2019 and decreased further to 8.35% in 2020. It then increased to 20.00% in 2021, declined

to 15.00% in 2022 and 2023, and increased to 20.00% in both 2024 and 2025. Litter cover was recorded as 0% at control sites in 2026. Litter cover was generally higher at control sites than at impact sites, except in 2020 and 2026.

Soil-crust cover was absent at both impact and control sites from 2017 to 2020 (Figure 10Figure 9). At impact sites, soil-crust cover increased to 1.00% in 2021 and reached a maximum of 7.00% in 2022. It then declined to 0% in 2023, 2024 and 2025 before increasing to 5.00% in 2026. Soil-crust cover remained at 0% at control sites throughout the survey period. Consequently, soil-crust cover was higher at impact sites than at control sites from 2021 onwards, except during the years when no soil crust was recorded at either site condition.

Scalded-surface cover was recorded as 0% at both impact and control sites from 2017 to 2021 (Figure 10Figure 9). At impact sites, scalded-surface cover remained at 0% throughout the survey period. At control sites, scalded-surface cover increased to 3.00% in 2022, declined to 0% in 2023, and increased slightly to 1.00% in both 2024 and 2025 before returning to 0% in 2026. Overall, scalded-surface cover was absent at impact sites and occurred only intermittently at control sites.

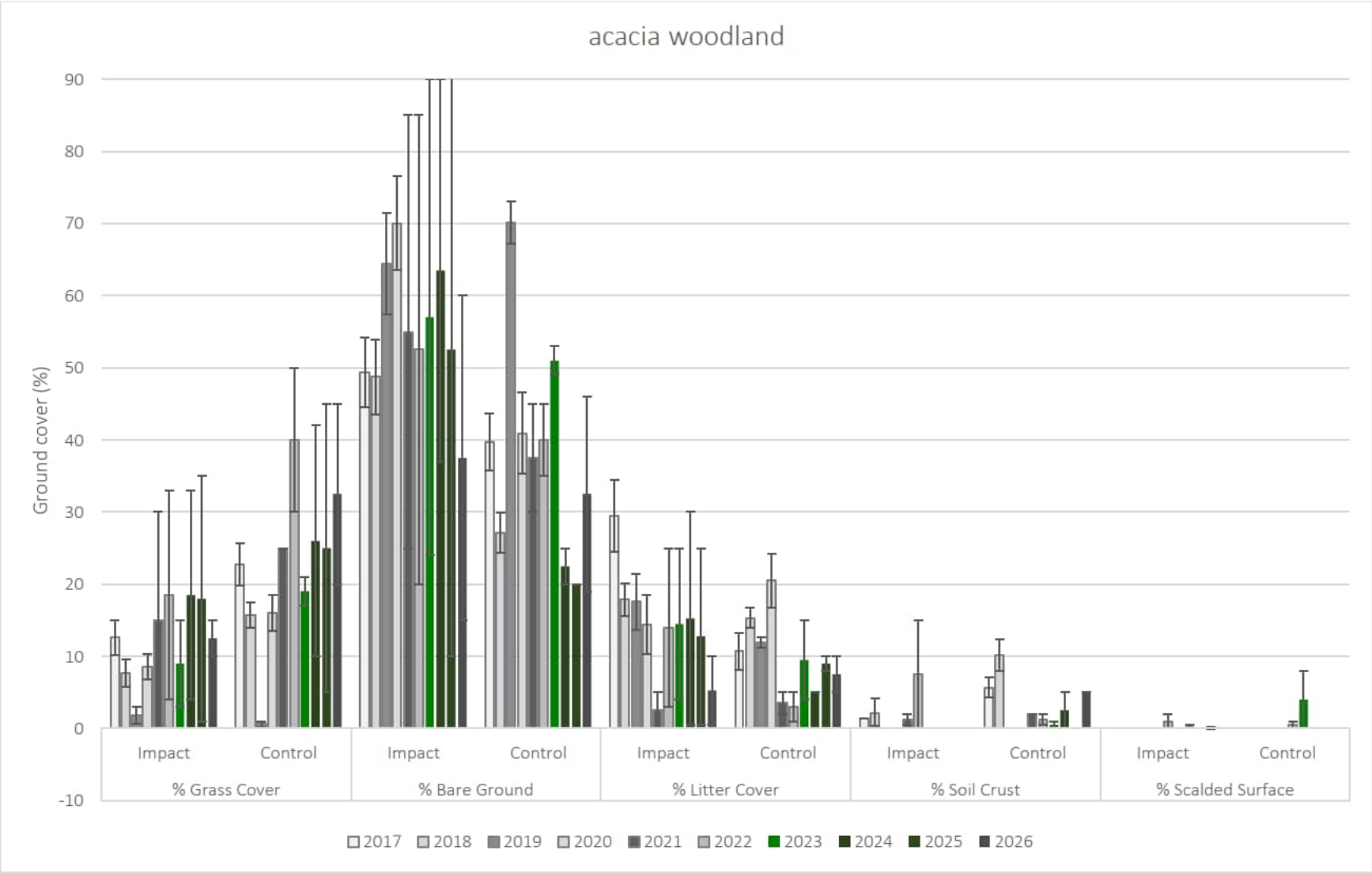


Figure 9: Mean percentage surface cover (%; $\pm$ SE) for Acacia Woodland across the 2017-2026 monitoring period

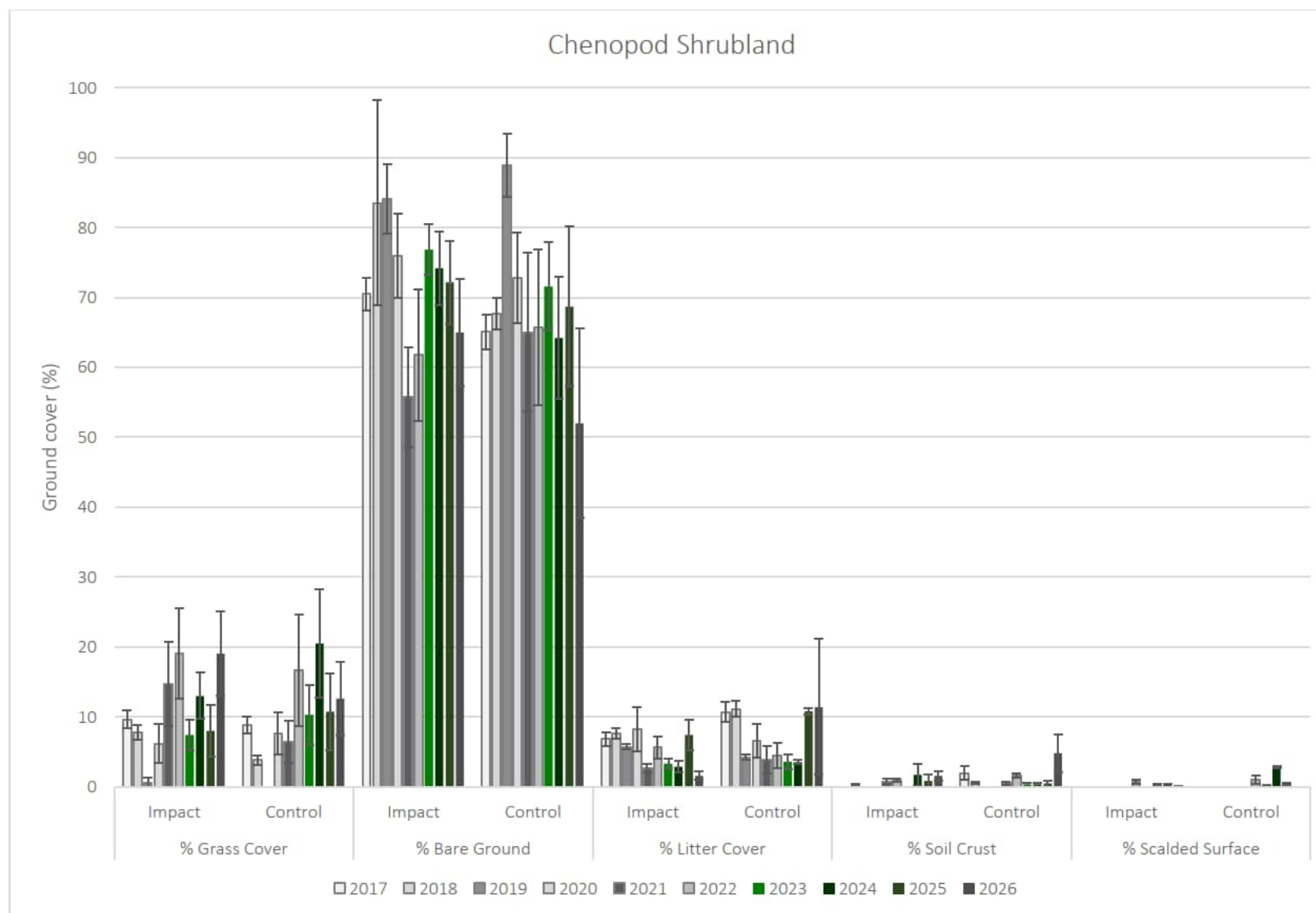


Figure 10: Mean percentage surface cover (%; $\pm$ SE) for Chenopod Shrubland across the 2017-2026 monitoring period

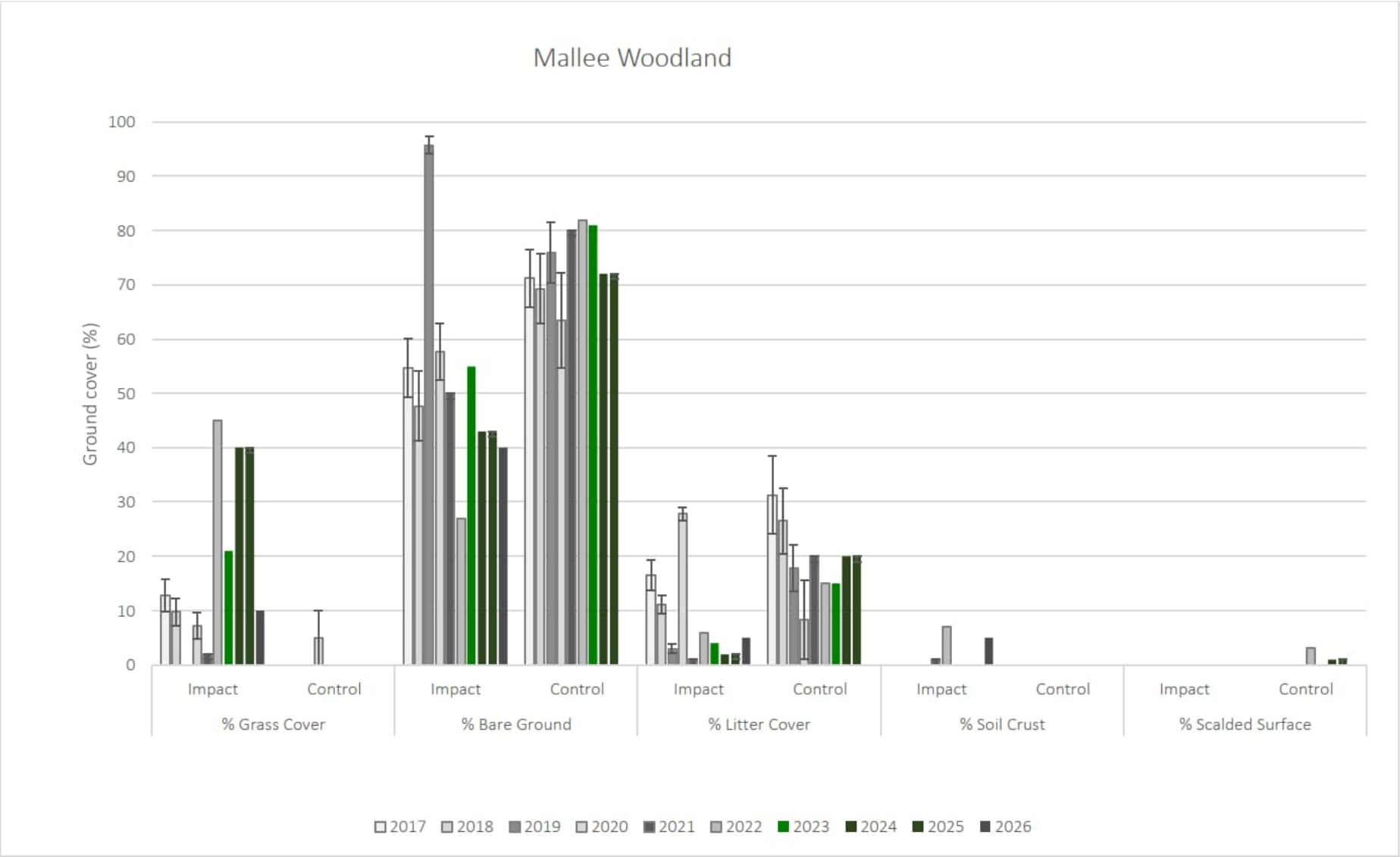


Figure 11: Mean percentage surface cover (%; $\pm$ SE) for Mallee Woodland across the 2017-2026 monitoring period

3.2. Weed monitoring sites

Each of the five monitoring sites consisted of five linear transects that were assessed for the presence of weeds. Two exotic species, *Cenchrus ciliaris* (buffel grass), and *Malvastrum americanum* var *americanum* were recorded in 2026. *Cenchrus ciliaris* was recorded within weed monitoring site 4 and this is the first record of this species since this survey began in 2017. Four individuals of *Malvastrum americanum* var *americanum* were recorded at weed monitoring site 3, and 200 individuals were recorded at weed monitoring site 4 (Figure 12). This is the first time since 2022 that *Malvastrum americanum* var *americanum* has been recorded at weed monitoring site 4.

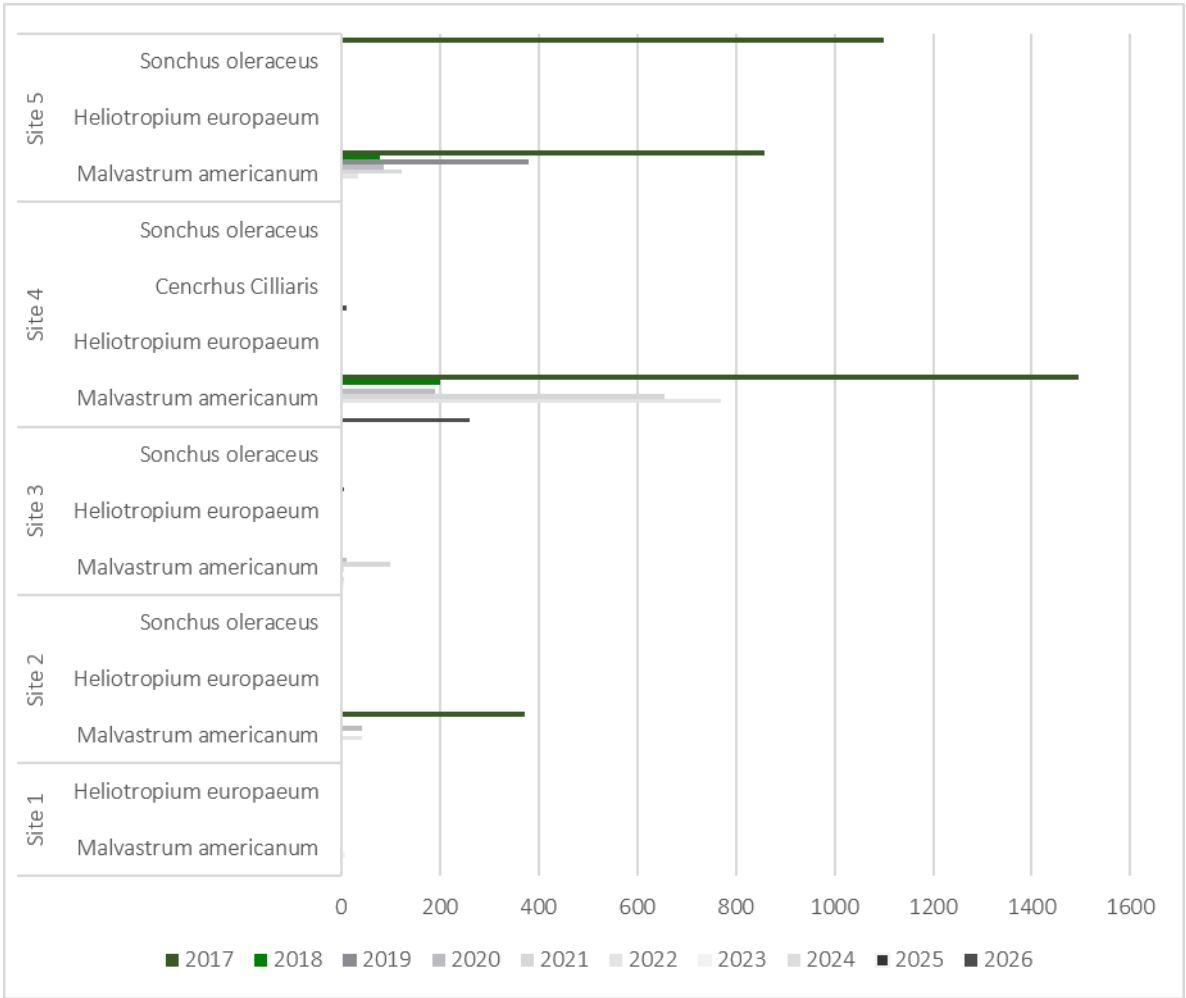


Figure 12: Weed abundance across the five weed monitoring locations 2017-2026 surveys

3.3. Incidental observations

During the 2026 survey, *Malvastrum americanum* was recorded at 15 of 17 of the monitoring sites, indicating that this species remains widely distributed in the area. Signs of rabbit activity were recorded at several sites, including warrens and scats at PH02, PH03, PH10, PH12, PH24 and PH25, while fox scats were observed at PH17 and PH25. Other observations, include incidental sightings of approximately four *Recurvirostra novaehollandiae* (red-necked avocet), seven *Malacorhynchus membranaceus* (pink-eared duck) and 15 *Tribonyx ventralis* (black-tailed native hen) within an ephemeral wetland adjacent to the access track, 12 km west of weed monitoring site 2.

Observational data for the floristic monitoring sites is present below in Table 5.

Table 5: Incidental observations recorded during the 2026 survey

Site	Vegetation type	Impact/control	Observations
PH01	Chenopod Shrubland	Impact	- Pitfall caps in good condition and pose no danger to wildlife. <i>Malvastrum americanum</i> present.
PH02	Acacia Woodland	Impact	<i>Malvastrum americanum</i> present. - Pitfall caps in good condition and pose no danger to wildlife. - Scattered dead acacias (approximately 10 individuals) throughout quadrat, historically dead. - Rabbit warren and scats.
PH03	Chenopod Shrubland	Impact	<i>Malvastrum americanum</i> present. - Pitfall caps in good condition and pose no danger to wildlife. - Rabbit warren and scats
PH06	Chenopod Shrubland	Control	<i>Malvastrum americanum</i> present. - Pitfall caps in good condition and pose no danger to wildlife.
PH07	Mallee Woodland	Control	Not monitored in 2026.
PH10	Chenopod Shrubland	Impact	<i>Malvastrum americanum</i> present. - Pitfall caps in good condition and pose no danger to wildlife. - Rabbit warren and scats 5-10 individuals of <i>Malurus leucopterus</i> (white-winged fairy-wren)
PH11	Chenopod Shrubland	Control	Not monitored in 2026.
PH12	Chenopod Shrubland	Impact	- Pitfall caps in good condition and pose no danger to wildlife. - Rabbit warren and scats
PH13	Chenopod Shrubland	Impact	<i>Malvastrum americanum</i> present. - Pitfall caps in good condition and pose no danger to wildlife.
PH15	Mallee Woodland	Impact	<i>Malvastrum americanum</i> present. - Pitfall caps in good condition and pose no danger to wildlife. - Minor water erosion/channels - Roosting/nesting (<i>Melopsittacus undulatus</i>) budgerigars in hollows in <i>Eucalyptus socialis</i>. - Basal regrowth of senescent <i>Eucalyptus socialis</i>
PH16	Chenopod Shrubland	Control	<i>Malvastrum americanum</i> present. - Pitfall caps in good condition and pose no danger to wildlife.

Site	Vegetation type	Impact/control	Observations
PH17	Chenopod Shrubland	Control	• <i>Malvastrum americanum</i> present. • Pitfall caps in good condition and pose no danger to wildlife. • Fox (<i>Vulpes vulpes</i>) scat
PH18	Chenopod Shrubland	Control	• <i>Malvastrum americanum</i> present. • Pitfall caps in good condition and pose no danger to wildlife.
PH19	Chenopod Shrubland	Impact	• Pitfall caps in good condition and pose no danger to wildlife.
PH20	Chenopod Shrubland	Control	Not monitored in 2026.
PH22	Acacia Woodland	Impact	• <i>Malvastrum americanum</i> present. • Pitfall caps in good condition and pose no danger to wildlife.
PH23	Chenopod Shrubland	Impact	• <i>Malvastrum americanum</i> present. • Pitfall caps in good condition and pose no danger to wildlife. • Stakes need to be redone for subplot and Jessup
PH24	Acacia Woodland	Control	• <i>Malvastrum americanum</i> present. • Pitfall caps in good condition and pose no danger to wildlife. • Rabbit warren and scats
PH25	Chenopod Shrubland	Control	• <i>Malvastrum americanum</i> present. Pitfall caps in good condition and pose no danger to wildlife. • Fox scat • Rabbit warren and scats
PH27	Acacia Woodland	Control	• <i>Malvastrum americanum</i> present. • Pitfall caps in good condition and pose no danger to wildlife.

4. Discussion

4.1. Floristic quadrat monitoring

The 2026 survey was undertaken over two field trips, with PH22 assessed during the first visit and the remaining floristic monitoring quadrats surveyed during the second visit in early August. Three permanent quadrats (PH07, PH11 and PH20) were inaccessible during the survey period and were therefore not monitored in 2026. Data from the 2026 survey were compared with previous years, and between impact and control sites where possible, to assess whether mining activities at Prominent Hill are influencing the surrounding vegetation communities.

4.1.1. Total species richness

In 2026, 148 species were recorded, comprising 145 native species and 3 exotic species. In 2026, 12 of the 17 floristic monitoring quadrats recorded at least one species that had not previously been identified within the vegetation monitoring sites. Statistical analysis did not identify a significant difference in total native species richness between impact and control sites for any of the vegetation types. In 2026, Acacia Woodland showed slightly higher mean total native species richness at impact sites than at control sites, with impact sites recording 47.5 species compared with 44.5 species at control sites. Chenopod Shrubland recorded higher mean native species richness at control sites than at impact sites, with 47.2 and 38.3 species, respectively. Mallee Woodland recorded 48 native species at the single impact site surveyed in 2026.

Overall, total species richness has been found to be variable since the survey began in 2017. Rainfall is limited in the South Australian Arid Lands (SAAL), and vegetation responses vary markedly following rainfall events. For example, total native species richness was lower across all treatment and vegetation types from 2017 to 2019, which corresponds with below average monthly and yearly rainfall. Therefore, the variation observed between years is considered to reflect variable rainfall and climatic condition in the SAAL which results in fluctuations in annual species diversity, and not a result of impacts from mining.

4.1.2. Native perennial species richness

In 2026, mean native perennial species richness was slightly higher at Acacia Woodland impact sites than at control sites (35.50 and 33.50 species, respectively); however, this difference was not statistically significant ($p = 0.864$), however the small sample size means that this result should be interpreted cautiously. Mallee Woodland recorded a mean of 38 perennial species at the single impact site surveyed, but no statistical comparison was possible because control-site data were unavailable. Chenopod Shrubland control sites supported significantly greater mean perennial species richness than impact sites (35.20 compared with 27.29 species; $p = 0.041$), indicating a difference of approximately eight species between conditions. The difference in species richness is likely due to pre-existing differences between control and impact sites and vegetation variation and fluctuations between survey years. Regardless, the difference between impact and control sites should be monitored closely in future surveys.

Across the monitoring period, perennial species richness generally declined between 2017 and 2020 in Acacia Woodland and Chenopod Shrubland, followed by recovery or stabilisation in subsequent years. Mallee Woodland impact sites increased substantially after 2019 and remained relatively stable from 2020 onwards. Overall, the results suggest that perennial species richness was generally maintained or

increased in recent years, although the significant difference recorded for Chenopod Shrubland indicates that impact and control responses were not consistent across vegetation types.

4.1.3. Total species cover

In 2026, total species cover showed no statistically significant differences between vegetation types or treatment types. Acacia Woodland recorded 74.5% cover at control sites and 62.5% at impact sites, while Chenopod Shrubland recorded 60.21% and 45.63%, respectively. However, neither difference was statistically significant, and the Acacia Woodland comparison was subject to considerable uncertainty because of the small sample size. Mallee Woodland recorded 68% native species cover at the single impact site surveyed, but no statistical comparison was possible because control-site data were unavailable.

Across all vegetation types and treatment types, a pattern of decreasing species cover can be seen from 2017-2019, before stabilisation and subsequent seasonal variability from 2020-2025. This corresponds to below average rainfall seen from 2017-2019, followed by more consistent rainfall from 2020-2026. Therefore, the variation in vegetation cover observed between years is considered to reflect variable rainfall and climatic condition in the SAAL which results in fluctuations in annual species diversity, and not a result of impacts from mining.

It is noted that in 2026 estimates of native species cover were high relative to previous surveys, which has potential to be due to observer bias and is an important factor in interpreting results of ecological surveys.

4.1.4. Native perennial species cover

In 2026, mean native perennial species cover was higher at control sites than at impact sites for both Acacia Woodland and Chenopod Shrubland. Acacia Woodland recorded 73.22% cover at control sites compared with 59.06% at impact sites, while Chenopod Shrubland recorded 58.85% and 41.37%, respectively. However, neither difference was statistically significant, with substantial uncertainty associated with the Acacia Woodland comparison due to the small sample size. Mallee Woodland recorded 54.2% perennial cover at the single impact site surveyed, but no statistical comparison was possible because control-site data were unavailable.

Similar to total species cover, perennial species cover data shows a pattern of decreasing species cover from 2017-2019, before stabilisation and subsequent seasonal variability from 2020-2025. Therefore, variation between survey years is considered to reflect variable rainfall and climatic condition in the SAAL which results in fluctuations in annual species diversity, and not a result of impacts from mining.

It is noted that in 2026 estimates of native species cover were high relative to previous surveys, which has potential to be due to observer bias and is an important factor in interpreting results of ecological surveys.

4.1.5. Native perennial density

In 2026, mean total, mature and juvenile perennial density was lower at impact sites than at control sites in both Acacia Woodland and Chenopod Shrubland. However, none of these differences were statistically significant. In Acacia Woodland, control sites recorded higher mean total stem density than impact sites (0.331 compared with 0.183 stems/m²), as well as higher mature density (0.293 compared with 0.160 stems/m²) and juvenile density (0.088 compared with 0.056 stems/m²). The corresponding *p*-values were 0.525, 0.566 and 0.684, respectively. Similarly, Chenopod Shrubland control sites recorded higher mean total stem density (0.349 compared with 0.283 stems/m²), mature density (0.302

compared with 0.227 stems/m²) and juvenile density (0.168 compared with 0.144 stems/m²), although none of these differences were statistically significant ($p = 0.461$, 0.235 and 0.608, respectively).

Perennial density varied between vegetation types and survey years, with the greatest changes observed in mature perennial density in Acacia Woodland and Chenopod Shrubland, and in juvenile perennial density at Acacia Woodland impact sites in 2026. Variability between survey years, may be a result of seasonal variation and observer bias.

4.1.6. Native recruitment

In 2026, subplot cover estimates and perennial recruitment showed no statistically significant differences between impact and control sites for either Acacia Woodland or Chenopod Shrubland. Acacia Woodland impact sites had slightly higher bare-ground cover, while control sites had higher grass cover, litter cover and perennial recruitment; however, all comparisons were non-significant and were subject to substantial uncertainty because of the very small sample size. Scalded-surface cover was 0% in both groups, and soil-crust cover had insufficient variation for statistical testing. Chenopod Shrubland impact sites recorded higher bare-ground cover, grass cover, scalded-surface cover and perennial recruitment, whereas control sites had higher litter and soil-crust cover. None of these differences were statistically significant, although the recruitment comparison showed a possible trend towards higher values at impact sites ($p = 0.078$). Overall, the results do not provide evidence of consistent differences in subplot ground-cover attributes or perennial recruitment between impact and control sites. The findings should nevertheless be interpreted cautiously because of small sample sizes, broad uncertainty around the estimates and limited variation in some variables.

Across the monitoring period, native perennial recruitment varied substantially between years and vegetation types. Recruitment was generally high in 2017, declined markedly during 2018 and 2019, and increased again in 2020, suggesting a strong response to interannual environmental variability. Recruitment continued to fluctuate between 2021 and 2026, with no consistent pattern separating impact and control sites. Overall, the variation in recruitment between years and the absence of statistically significant differences in 2026 suggest that recruitment patterns are likely influenced by annual rainfall and climatic conditions, as well as site-level variability, rather than impacts from mining.

4.1.7. Subplot monitoring

Subplot-level monitoring showed variation in ground-layer attributes among vegetation types and treatment categories, although most variables did not differ significantly. Bare ground was generally high across the monitoring sites, particularly in Chenopod Shrubland, while litter and soil crust varied more noticeably between impact and control areas. In Acacia Woodland, impact sites tended to have more bare ground and less grass cover, litter and soil crust than control sites. In Chenopod Shrubland, bare ground was also higher at impact sites, grass cover was slightly higher, and litter cover was lower; scalded surface was only recorded at impact sites. Mallee Woodland again had only impact-site data available in 2026. Therefore, it is reasonable to attribute variation between sites to random effects rather than impacts from Prominent Hill Mining activities.

Perennial recruitment within the subplots also varied between vegetation types and treatments. In Acacia Woodland, perennial recruitment was lower at impact sites than at control sites, whereas in Chenopod Shrubland it was higher at impact sites. Mallee Woodland had only impact-site data available in 2026. Overall, statistical analysis did not indicate a significant effect of vegetation type on perennial recruitment in the subplots. Therefore, it is reasonable to attribute variation between sites to random effects rather than impacts from Prominent Hill Mining activities.

4.1.8. Weed monitoring sites

The 2026 weed monitoring results indicate that exotic species remain present within the monitoring area, although overall weed occurrence was limited to two species. The detection of *Cenchrus ciliaris* at weed monitoring site 4 is notable as the first record of this species since monitoring began in 2017. While it was only detected at a single site, its presence should be monitored as it is a highly invasive grass with the potential to establish and spread. *Malvastrum americanum* var. *americanum* was recorded at two monitoring sites in 2026, with a small number of individuals at site 3 and a much larger population at site 4. This is the first record of the species at site 4 since 2022, suggesting that it may persist in the area at low levels and re-emerge under favourable conditions. No other weeds were detected within the monitoring sites, which may be attributed to the confirmed management of weeds within the mine site and along the haul road as well as within the village area.

It is of note that *Malvastrum americanum* var. *americanum* was identified in 15 out of 17 floristic monitoring quadrats. This species is listed as a 'possible weed' on the South Australian Census, but it is not a 'declared weed' (eFlora 2020, PIRSA 2020). It should also be noted that both eFloraSA, the electronic taxonomic key of the Flora of South Australia and PlantNET, the online taxonomic key of the Flora of New South Wales, identify *Malvastrum americanum* var. *americanum* as naturalised (eFloraSA 2020, PlantNET 2023).

4.2. Survey limitations and recommendations

Observer bias and differences in data collection between survey years may reflect changes in survey teams, seasonal conditions and site accessibility, rather than actual changes in vegetation condition.

In addition, the 2026 survey found pickets missing from several floristic monitoring quadrats. Although GPS units were used to assist with site relocation, positional accuracy in remote areas can be affected by equipment limitations, satellite reception and topographical features, especially in a remote location such as Prominent Hill. Inconsistencies between the relocation and demarcation of vegetation monitoring sites, Jessup transects and/ or subplots, may have contributed to variability in the results between survey years. Therefore, it is recommended that markers (i.e. pickets) for the floristic monitoring quadrats, Jessup transects and subplot are maintained prior to, or as part of, the annual ecological survey.

The statistical results should be interpreted cautiously because sample sizes differed substantially among vegetation types. Acacia Woodland had only two impact sites and two control sites, resulting in low statistical power and considerable uncertainty around the estimated differences. Mallee Woodland had only one impact site and no control sites available in 2026; therefore, statistical comparisons between conditions could not be conducted for this vegetation type. Chenopod Shrubland had greater replication than Acacia Woodland and Mallee Woodland, but the sample size was still relatively small and natural variability between sites may have limited the ability to detect differences. The results of the statistical analysis should therefore be considered alongside the observed means, effect sizes, confidence intervals and ecological trends, rather than relying on statistical significance alone.

5. Assessment against outcome measurement criteria

A summary of the relevant compliance and outcome criteria contained within the PEPR is provided overleaf in Table 6, demonstrates that the objectives of this survey, as presented in Section 1.3, have been met.

Table 6: Summary of measurement criteria (Reference: Table 7-12 in the 2022 PEPR) assessed against the results of the ecological assessment in 2026

Grouped condition number	Environmental outcome	OMC details	Statement addressing the OMC	Outcome achievement	LIC	Statement addressing the LIC
GC12	No loss of abundance or diversity of native vegetation, or reduction in habitat quality, on or off the Mining and or miscellaneous purposes lease areas during construction, operation as a result of mining activities unless prior approval under relevant legislation is obtained and environmental offsets are approved and in place.	10 x 2 m sub plot within vegetation quadrat. For each 1 x 1 m unit of sub-plot: - o Estimate of % grass cover (ephemerals, annuals) - o Estimate of % bare ground - o Estimate of % litter cover within the plot - o Estimate of % surface crust - o Counts of recruits (all shrubs) to provide recruitment score - o Long lived perennials (over and understory via species abundance counts (density) for both juveniles and adults. Panoramic photographs collected to aid in assessment of vegetation cover for recruitment. Observational data to be collected at all sites including (but not limited to) vehicle tracks, erosion, vegetation clearing, distance to mine site, light, dust, inappropriate access, feral animals and weeds. GIS output of approved clearance boundary and actual clearance boundary.	The species diversity, vegetation cover and the percentage of species recruiting in 2026 was stable compared to 2024-2025, and high compared to the historical data (2017-2024). Rabbit activity was observed consistent across all sites for all three vegetation types. ELA cannot confirm an approved Native Vegetation Management Plan and SEB Offset in place. Our results are not relevant to any clearance, unapproved or otherwise and the survey work did not include review of clearance boundaries. We understand BHP include this work in their annual reporting.	Approved Native Vegetation Management Plan and SEB Offset in place.	Indicators of habitat degradation, including: - o Reduction of perennial species abundance (counts) at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods. - o Suppression of recruitment indicated by a reduction in recruitment index scores at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods. - o An increase in bare ground and/ or scald/ erosion % cover at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods. If leading indicator triggered, implement targeted threatened bird surveys to confirm ongoing presence of thick-billed grasswren and chestnut-breasted whiteface within impacted sites.	The following observations have been made in relation to the leading indicator criteria: 1. The average total species richness, and perennial species richness was stable in 2026. Native vegetation cover (abundance) has remained consistent over the 2021-2026 period. 2. The proportion of species recruiting is similar at impact and control sites across all three vegetation types. 3. The amount of grass cover at all sites was overall stable in 2026. 4. Bare ground cover has decreased in 2026 at all sites. Our results are not relevant to any clearance, unapproved or otherwise and the survey work did not include review of clearance boundaries. We understand BHP include this work in their annual reporting.
GC13 GC14	Environmental offsets are approved and in place for all clearance of native vegetation	As above	As above	As above	Reduction of perennial species abundance (counts) at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods. Suppression of recruitment indicated by a reduction in recruitment index scores at impact sites without a corresponding reduction at control sites over three consecutive monitoring periods for Chenopod Shrubland. An increase in bare ground and/ or scald/ erosion % cover at impact sites without a corresponding reduction at control over three consecutive monitoring periods. If leading indicator triggered, further assessment of detected impacts vs pre-mine condition by comparison with sites outside of SEB area required to determine non-compliance with lease condition. Annual review of vegetation clearance confirms all clearance has been approved.	As above
GC64	No introduction of new species of weeds, plant pathogens or pests (including feral animals), nor sustained increase in abundance of existing weed or pest species in the licence area compared to adjoining land as a result of mining operations.	Records of counts and identification of any declared weed infestations observed along transects at each location. Records of weed control actions implemented (where relevant). Records of feral animal observations, trapping and capture rates.	Weed monitoring was completed in and outside of the ML. <i>Cenchrus ciliaris</i> was recorded within weed monitoring site 4 and this is the first record of this species since this survey began in 2017. <i>Malvastrum americanum</i> var <i>americanum</i> was observed in higher abundance at all sites; however, this perceived increase was not statistically significant.	Weed infestations are recorded, treated and monitoring for ongoing management requirements	Annual review of the weed survey and management register (results of field monitoring and visual observations) considering trends that could indicate population increase or new weed species. Quarterly review of cat sightings and trapping register considering trends that could indicate population increase and requirement for increase in trapping program.	<i>Cenchrus ciliaris</i> was recorded within weed monitoring site 4 and this is the first record of this species since this survey began in 2017.

6. Conclusion

The 2026 ecological assessment at Prominent Hill does not indicate measurable loss of abundance or diversity of native vegetation, or reduction in habitat quality, the vegetation condition observed within the impact sites is in most cases, consistent with the surrounding control sites. However, it was found that total native species richness in sites supporting Chenopod Shrubland vegetation type was significantly different between impact and control sites. The difference in species richness is likely due to pre-existing differences between control and impact sites and vegetation variation and fluctuations between survey years. Regardless, this should be monitored in future surveys.

In 2026, weed monitoring found *Cenchrus ciliaris* was recorded within weed monitoring site 4 and this is the first record of this species since this survey began in 2017. This is a highly invasive grass species and should be monitored closely.

The results of the Autumn 2026 ecological survey do not provide evidence that the Prominent Hill mine is adversely affecting native vegetation. Descriptive ecological data indicate that native vegetation abundance, species richness, diversity and recruitment at impact sites followed patterns broadly consistent with those observed at control sites across the surveyed vegetation types. The statistical analyses generally supported this interpretation; however, the results should not be considered in isolation because they may be influenced by climatic variability, observer bias, pre-existing differences between sites and limited replication, particularly for Acacia Woodland and Mallee Woodland.

7. References

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- ELA (2022) Prominent Hill Annual Vegetation and Habitat Compliance Monitoring. Prepared for OZ Minerals.
- ELA (2023) Prominent Hill Annual Vegetation and Habitat Compliance Monitoring. Prepared for OZ Minerals.
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Appendix A - 2026 species matrix

Table 7: Vegetative life stage key

Abbreviation	Definition
B	Budding
F	Flowering
I	Immature fruit
M	Mature fruit
V	Vegetative
D	Dead

Appendix B - Floristic monitoring sites and panoramic photographs

Site ID	Vegetation Community	Type	Northwest corner	
			Latitude	Longitude
PH01	Chenopod Shrubland	Impact	29.71758652S	135.52812195E

2019



2020



2021



2022



2023



2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH02	Acacia Woodland	Impact	29.71266747S	135.59562683E

2019



2020



2021



2022



2023



2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH03	Chenopod Shrubland	Impact	29.69483376S	135.57168579E
2019				
				
2020				
				
2021				
				
2022				
				
2023				
				

2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North-west corner	
			Latitude	Longitude
PH06	Chenopod Shrubland	Control	29.74950446S	135.52990631E
2019				
				
2020				
				
2021				
				
2022				
				
2023				
				

2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH07	Mallee Woodland	Control	29.68139458S	135.57141113E

2019



2020



2021



2022



2023



2024



2025



2026 - N/A

Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH10	Chenopod Shrubland	Impact	29.701975S	135.593815E
2019				
				
2020				
				
2021				
				
2022				
				
2023				
				

2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH11	Chenopod Shrubland	Control	29.73239136S	135.51400757E

2019



2020



2021



2022



2023



2024



2025



2026 – N/A

Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH12	Chenopod Shrubland	Impact	29.72888651S	135.55339703E
2019				
				
2020				
				
2021				
				
2022				
				
2023				
				

2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH13	Chenopod Shrubland	Impact	29.74042511S	135.59289551E

2019



2020



2021



2022



2023



2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH15	Mallee Woodland	Impact	29.69200524S	135.59357799E
2019				
				
2020				
				
2021				
				
2022				
				
2023				
				

2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH16	Chenopod Shrubland	Control	29.71088982S	135.61737061E
2019				
				
2020				
				
2021				
				
2022				
				
2023				
				

2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH18	Chenopod Shrubland	Control	29.68750763S	135.55403137E

2019



2020



2021



2022



2023



2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH19	Chenopod Shrubland	Impact	29.71213913S	135.55108643E
2019				
				
2020				
				
2021				
				
2022				
				
2023				
				

2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH20	Chenopod Shrubland	Control	29.68099976S	135.61598206E

2019



2020



2021



2022



2023



2024



2025



2026 – Not surveyed in 2026

Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH22	<i>Acacia</i> Woodland	Impact	29.7509861S	135.55839539E

2019



2020



2021



2022



2023



2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH23	Chenopod Shrubland	Impact	29.70710564S	123.55673218E

2019



2020



2021



2022



2023



2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH24	Acacia Woodland	Control	29.76878357S	135.53315735E
2019				
				
2020				
				
2021				
				
2022				
				
2023				
				
2024				
				

2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH25	Chenopod Shrubland	Control	29.75158433S	135.52481204E
2019				
				
2020				
				
2021				
				
2022				
				
2023				
				

2024



2025



2026a



2026b



Site ID	Vegetation Community	Type	North west corner	
			Latitude	Longitude
PH27	<i>Acacia</i> Woodland	Control	29.75692749S	135.55107117E

2019



2020



2021



2022



2023



2024



2025



2026a



2026b



