Melbourne Strategic Assessment

Population viability analysis models for threatened species: Version 1.0

Tracey Regan, Matthew Bruce, Khorloo Batpurev, Brad Farmilo, Michael Scroggie, Billy Geary and Natasha Cadenhead

June 2021

Arthur Rylah Institute for environmental research

VICTORIA State Government | Environment, Land, Water and Planning

Accessibility

If you would like to receive this publication in an alternative format, please telephone the DELWP Customer Service Centre on 136 186, email customer.service@delwp.vic.gov.au or contact us via the National Relay Service on 133 677 or www.relayservice.com.au. This document is also available on the internet at www.delwp.vic.gov.au

Melbourne Strategic Assessment

Population viability analysis models for threatened species: Version 1.0

Tracey Regan[1], Matthew Bruce[1], Khorloo Batpurev[1], Brad Farmilo[1], Michael Scroggie[1], Billy Geary[2] and Natasha Cadenhead[3]

[1]Arthur Rylah Institute for Environmental Research, The Department of Environment, Land, Water and Planning,123 Brown Street, Heidelberg, Victoria 3084

[2]Biodiversity, Strategy and Knowledge, Biodiversity Division, The Department of Environment, Land, Water and Planning, 8 Nicholson Street, East Melbourne, Victoria, 3002

[3]School of Biosciences, The University of Melbourne, Parkville, Victoria 3010

Acknowledgements

Contents

Unofficial

Summary

Context

The Melbourne Strategic Assessment (MSA), is committed to a range of conservation outcomes including the persistence of all EPBC listed threatened species impacted by the expansion of the Melbourne's urban growth boundary. The MSA ecological program focuses on providing the evidence-base to support management choices to achieve the conservation commitments agreed to under the MSA program.

Aims

- To develop population viability analysis models (PVA) for nine EPBC listed species that are part of the MSA program that integrates current science, data and expert judgement to explore the relative merits of alternative management actions on the persistence of each species.

- Identify key uncertainties relating to the management of each species to assist in prioritising research

Approach

Species specific PVA models were developed using a consistent set of steps that include:

Step 1 Desktop review: Identify existing models, data sources and scientific literature relevant to the species.

Step 2 Meet with experts: Develop a conceptual model of the life history of the species, identify key data sets, suite of management options and agreed approach to proceed with model development.

Step 3 Draft quantitative PVA model: Translate conceptual model into a draft quantitative PVA model.

Step 4 Expert elicitation workshop: Elicit expert judgements on all model parameters using a structured approach.

Step 5 PVA model development: Develop species specific PVA models. Evaluate all management scenarios and perform sensitivity analysis.

Step 6 Review Process: Expert review of model structure, outputs and interpretation.

Results

All grassland species benefit from some form of control of grassland biomass (hereafter: biomass control) but there are differences in the preferred biomass control action across species. Golden Sun Moth and Striped Legless Lizard preferred grazing regimes over burning regimes, while the grassland plants preferred short or medium fire intervals depending on the species. Additional actions, in conjunction with biomass control, have species specific benefits. These actions include watering during drought, excluding rabbits and removal of perches to reduce predation by birds (i.e. Golden Sun Moth).

Sensitivity analysis revealed key inputs that influence model outcomes the most. While these parameters varied across species, information about survival rates under different biomass conditions for grassland species was common across species. Collecting information about key demographic processes and time series of population sizes, particularly for models that were largely driven by expert judgement is needed.

Conclusions

The PVA models developed for MSA species provide a strong foundation for on-going research to inform our understanding of extinction risk of these species. They also provide a valuable on-going resource to inform and improve management of the species over time, promote evidence-based decision making and contribute to the MSA meeting its ecological commitments.

Further refinements to models could be made by exploring a greater range of management options. The models provide estimates of the consequences of different actions for each species, but trade-offs may be required when planning management for multiple species. We recommend using Structured Decision Making as a process for developing guidelines for appropriate use of these models in practice. We encourage users to explore the model outputs through a web application and provide feedback so that future iterations of the models can be improved and informed by the most up to date information.

1 Introduction

Under the Melbourne Strategic Assessment (MSA), the Victorian Government is committed to a range of conservation outcomes including the persistence of all EPBC listed threatened species impacted by the expansion of the Melbourne's urban growth boundary. The MSA ecological program focuses on the science and research required to provide an evidence-base for management choices for threatened species' to achieve desired conservation outcomes.

As part of the MSA ecological program, population viability analysis (PVA) models were developed for each threatened species in the MSA. These models were used to evaluate the relative effectiveness of alternative management options. PVA models are quantitative models that integrate information on a species' life history, demographic processes and the influence of threats and stochastic events to predict the long-term persistence of a species. They are particularly powerful tools for systematically evaluating alternative management actions in terms of their contribution to a species' long-term persistence and for identifying key knowledge gaps for future research through sensitivity analysis.

The development of species' specific PVAs offers a range of benefits including:

- Evaluating alternative management actions to promote successful delivery of program outcome commitments

- Developing a shared understanding of the life history, population dynamics and how threats and management influence demographic processes

- Integrating various sources and forms of information and data, including empirical data and expert judgement

- Providing a transparent repository of information that can be updated over time and encourage contribution from the scientific community

- Providing a transparent exploration of the trade-offs between the benefits of alternative management actions

- Identifying key uncertainties relating to the management of species thus assist in prioritising research.

This report outlines the development of PVA models for each of the nine MSA species. Each species has a dedicated chapter that includes an expert derived conceptual model of the life history of the species identifying how external factors affect demographic processes, details of the structure of the PVA model, key assumptions, parameter estimates, alternative management actions investigated, results, sensitivity analysis and discussion. The final chapter synthesizes the results from all the PVA models and describes a structured decision-making process for using the models to inform decision making within particular decision contexts. The remainder of this chapter outlines the scope of this project, introduces the species, modelling approach and management options common to all species.

The scope of this work is set by several factors:

- Conservation outcomes for the MSA have been approved by the Commonwealth and are outlined in the Program Report (DPCD 2009)

- The Monitoring and Reporting Framework (DELWP, 2015) for the MSA outlines the program's outcomes and how these will be measured

- The spatial application of the models is defined as the MSA conservation and management areas identified in the Biodiversity Conservation Strategy (DEPI, 2013), Program Report (DPCD, 2009), and Sub-regional Species Strategies.

- Management actions will be restricted to those that can be feasibly delivered under the program.

1.1 Species

The MSA focuses on nine threatened species, five plants and four animals (Table 1.1). The species occur in the Western Grasslands Reserve (WGR) and a network of conservation reserves within Melbourne's Urban Growth Boundary and Conservation Areas on the Victorian Volcanic Plains.

Table 1.1: Threatened species in the MSA program.

Common name	Scientific name	Description
Spiny Rice-flower (SRF)	*Pimelea spinescens* subsp. *spinescens*	Small shrub
Button Wrinklewort (BWW)	*Rutodosis leptorhynchoides*	Perennial daisy
Matted Flax-lily (MFL)	*Dianella amoena*	Perennial lily
Small Golden Moths Orchid (SGMO)	*Diuris basaltica*	Deciduous, perennial orchid
Large-fruit Groundsel (LFG)	*Senecio macrocarpus*	Perennial daisy
Golden Sun Moth (GSM)	*Synemon plana*	Insect
Southern Brown Bandicoot (eastern) (SBB)	*Isoodon oblesulus oblesulus*	Mammal
Striped Legless Lizard (SLL)	*Delma impar*	Reptile
Growling Grass Frog (GGF)	*Litoria raniformis*	Amphibian

1.2 Methods

When developing the PVA models for each species, we followed six steps. The aim was to promote consistency across models and to improve efficiencies in model development. The following steps were followed for all species except the Growling Grass Frog (GGF). Model development for GGF commenced several years prior to other MSA species and a different modelling approach (Bayesian stochastic patch occupancy model) was developed to explore management options for this species.

Step 1: Desktop review and draft conceptual model

A desktop review was conducted to identify existing models, data sources and scientific literature relevant to the species and model development. Empirical data and literature from similar species were also reviewed for suitability. Existing PVA models were assessed against the needs of the MSA program and their ability to be adapted to meet the MSA program needs. A draft conceptual model was developed for each species to illustrate DELWP's current understanding of the life history of the species, identify key ecological components and processes, how different threats operate and which components and processes they target, and how management interventions can influence the system. For some species, a yearly timeline was also created to identity when important processes occur. These draft conceptual models played a key role in facilitating discussions with experts and in developing a shared understanding of the life histories of the species.

Step 2: Meeting with experts

An initial meeting was held with species experts where we presented the scope and context of the MSA model development, discussed the outcomes from the desktop review, potential management options for evaluation, key gaps in knowledge of the species and its requirements, and identified additional data sources not identified in the review. This meeting was also used to discuss, update and agree on the conceptual model for the species as these models functioned as the first step in the development of a quantitative model.

Step 3: Draft quantitative PVA model

After the initial meeting with experts, the conceptual model was updated and translated into a draft quantitative PVA model. Data sources to parameterise the model were reviewed for suitability in the PVA.

Step 4: Expert elicitation workshop

A second workshop with species experts discussed model structure and elicited expert judgements on all model parameters. The elicitation was done in a structured way. Each expert was asked individually to

estimate parameters by providing an upper value, a lower value, a best guess and level of confidence such that the best guess was captured within their upper and lower bounds. All model parameters were elicited, even in cases where empirical data were available.

Step 5: PVA model development

We adapted existing PVA models where available. When assessing data sources to use in the models we used general rules of thumb. Firstly, we preferred to use empirical data of the species in the MSA management area. If unavailable, we then preferred to use relevant empirical data of the species from outside the management area. Our third preference was to use relevant empirical data from species with similar life histories in or outside the management area, and lastly, we used expert judgements only when empirical data was unavailable or insufficient to provide an adequate estimate for model parameters. Data availability was variable across species. For instance, Growling Grass Frog is predominantly data driven while the Small Golden Moths Orchid is purely expert driven. All other models have some combination of expert judgement and existing data.

To ensure models did not result in unrealistic outcomes (i.e. unexpected exponential growth or unrealistic population crashes) parameter values were adjusted within the plausible range of parameter values estimated from the available empirical data and expert judgements. Additional consultation with experts occurred out of workshop sessions where required. Model behaviour and preliminary results were discussed with at least one species expert to ensure realistic model behaviour as part of the model calibration process.

Step 6: Review Process

The models and evaluation scenarios were sent to species experts for review and comment. The review process ensured the information extracted from the workshops was interpreted and integrated correctly into the model, and the model was developed in a way that is consistent with experts understanding of the species.

1.2.1 Population Models

All models except for GGF were developed in RAMAS GIS v 6.0 (Akçakaya and Root, 2013). For specifics of GGF model see chapter 10. All RAMAS models were a-spatial except for the Southern Brown Bandicoot where a spatially explicit model was necessary due to the spatial dimension of alternative management actions investigated (i.e. creation of habitat corridors) and the species being highly mobile.

All models include both environmental and demographic stochasticity. Environmental stochasticity is the year to year variation in vital rates due to environmental conditions (i.e. weather, food resources etc). Demographic stochasticity is the variation in population growth that arises from the probabilistic nature of individual births and deaths. Demographic stochasticity is particularly important to account for in small populations, where species are more susceptible to extinction due to probabilistic births and deaths even when mean growth rates are positive (Akçakaya, 1991).

For some grassland species, a portion of the environmental stochasticity was modelled explicitly. Many grassland flora experience a germination pulse when there is above average rainfall in autumn, referred to as an "autumn break". This was modelled as a probabilistic event, implemented in the models as a reverse catastrophe (i.e. a probabilistic event inducing a positive effect). The average autumn rainfall between 1942-2016 is 130mm based on BOM data at Laverton (station ID: 012305). Above average rainfall was estimated to occur with a probability of 0.125, which equates to an average of once every eight years. Species responses to an autumn break varied between species and are specified in the relevant species chapters.

All models were run for 50 years into the future with 1000 simulations. Results were summarised as expected minimum abundance (EMA). This is calculated as the average of all the minimum abundances for each 50-year trajectory over the 1000 simulations. This summary metric is useful when evaluating management options for threatened species because it encapsulates stochasticity that is a key driver of extinction in species (McCarthy and Thompson 2001).

1.2.2 Expert elicitation

The lack of empirical data on the species life history and response to management necessitated the use of expert judgments to fill knowledge gaps. The use of expert judgment has been widely used in conservation science and environmental policy when empirical data are scarce or unavailable (Martin et al. 2012). However expert knowledge is plagued by cognitive and motivational biases, which can lead to poor inferences (Ludwig et al. 2001). Adopting formal structured elicitation approaches can help avoid some of these issues and improve the quality and accuracy of elicited knowledge (Burgman et al. 2011). This is done by treating expert elicitation as formal data acquisition, similar to field data collection, and subject it to the

same scrutiny to ensure that uncertainty is quantified, biases are minimized, and assumptions and reasoning are explicit and transparent (O'Hagan et al. 2006, McBride and Burgman 2011, Martin et al. 2012).

We elicited judgements from species' experts. The number of experts ranged between three and six depending on the species. The elicitation was done in a structured way where each expert was asked individually to estimate parameters by providing an upper value, a lower value, a best guess and level of confidence such that the best guess was captured within their upper and lower bounds. All model parameters were elicited, even in cases where empirical data were available. However, expert judgements were only used to parameterise models when empirical data was lacking and with the view that they will be updated over time when new information becomes available. Judgements for each model parameter were averaged across experts. Expert elicited bounds on parameter values were used to help calibrate the models as outlined in Step 5 above. The elicitation was conducted at face-to-face workshops because they improve the elicitation process by:

- resolving linguistic uncertainty and reducing misinterpretations
- sharing additional information
- allowing greater engagement and participation with the process.
- enabling experts to justify and explain opinions
- hearing counter arguments that may reduce overconfidence
- allowing the facilitator to monitor involvement and guide discussion

1.3 Management scenarios

1.3.1 Biomass control

In grasslands, biomass control is a key management action achieved through either burning or grazing. Biomass accumulates at different rates depending on the method of control (Figure 1.1). While cover of biomass is lower immediately after burning, the rate of accumulation (approximated here by percent cover of all vegetation), increases more rapidly after fire than when grazing is used to control biomass. We explored four common scenarios of burning that focus on different seasons and frequency (Table 1.2). These scenarios were applied to all grassland species unless otherwise stated within the species chapter. Grazing as a management action is explored for some species (i.e. Golden Sun Moth and Striped Legless Lizard). For other species, grazing was deemed by the experts to be always unsuitable and thus out of scope for this study. Grazing scenarios tended to be focused on frequency and intensity. Specific grazing scenarios explored using the models are outlined in chapters 7 and 9.

Weed management is vital for managing grasslands. We did not investigate different weed management options. Instead weed management was assumed to be part of each the biomass control actions. Weed management included targeted spot spraying of broadleaf and grassy weeds timed directly after burning, or when deemed appropriate during grazing.
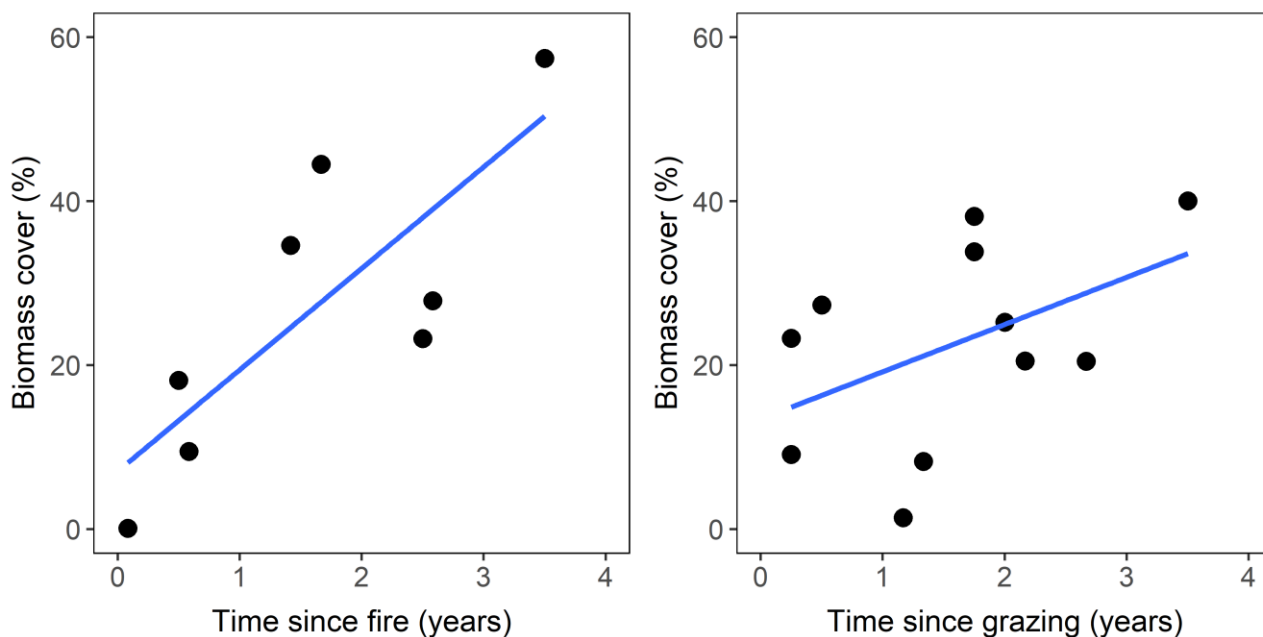
Figure 1.1: Relationship between time since biomass control (years) and biomass accumulation (%cover).

Table 1.2: Alternative burning scenarios explored in the models including interval, season and frequency

Scenario	Description	Fire interval	Season	Proportion of spring fires	Burn window	Median interval
1	No Management	No fire management	-	-	-	-
2	Short fire interval, Spring/Autumn	Short	Autumn, Spring	0.25	1 to 3 years	2 years
3	Medium fire interval, Autumn	Medium	Autumn	0	2 to 4 years	3 years
4	Medium fire interval, Spring	Medium	Spring	1	2 to 4 years	3 years
5	Long fire interval, Spring/Autumn	Long	Autumn, Spring	0.5	3 to 7 years	5 years

1.4 Sensitivity Analysis

Sensitivity analysis is an exploration of how the uncertainty in model parameters influences model outcomes. It is useful for highlighting how robust the model outputs are to uncertainty. If the model is sensitive to parameters that are highly uncertain, particularly if small changes induce large changes in model outcomes, or potentially change the ranking of management actions, then this suggests effort attaining more accurate data for those parameters is warranted.

We explored three types of sensitivity analysis as listed below. Each type of sensitivity analysis provides useful information regarding the behaviour of the model with respect to the model structure and the underlying data.

1. Deterministic sensitivity analysis is an analysis of the vital rates (i.e. survival and fecundity) that underpin the population model and highlights the contribution of each vital rate to the overall growth

Unofficial

rate. The metrics reported are elasticities that represent the proportional contribution of each vital rate to the population growth rate.

2. Stochastic sensitivity analysis is done through simulation. It explores how small changes in vital rates and other input parameters influence model outputs. The model is deemed sensitive to a parameter if a particular percent change in the parameters (i.e. ±10%) causes a larger percent change in the model outcome.

3. Scenario sensitivity analysis explores whether sensitive parameters and other model assumptions change the rank order of different model scenarios. This analysis tests the robustness of the model outcomes with respect to the ranking of alternative actions given uncertainty in the underlying parameters.

Specifics of the sensitivity analysis for each species model are available in the respective chapters.

2 Spiny Rice-flower (*Pimelea spinescens subsp. spinescens*)

2.1 Background

Spiny Rice-flower (*Pimelea spinescens* subsp. *spinescens*) is a long-lived (>100 years), perennial subshrub endemic to Victoria. It is largely dioecious, with separate male and female plants, and occasional hermaphrodites. It grows up to 30cm in height and has small, whiteish flowers. Germination generally occurs from winter to spring, and low competition induced by frequent burning combined with good seasonal rainfall will stimulate good levels of recruitment. Plants reach reproductive age at roughly two years and reproduce throughout their entire adult life. Once established, the Spiny Rice-flower is resilient to fire due to having a large tap-root which is protected and can resprout after fire (Mueck 2000, Carter and Walsh 2006).

Spiny Rice-flower (SRF) is listed as critically endangered under the EPBC act (1999) and vulnerable under the FFG act (1988). The species is confined to lowland grassy habitats in western and northern Victoria. The main threats to the species include habitat loss, competitive biomass including weed invasion, intense or prolonged grazing and inappropriate fire regimes. Approximately 80,000 plants occur in about 224 populations across its distribution (Carter and Walsh 2006). Within the MSA management area, SRF occurs in the western grassland reserve and conservation areas 1,2,3p, 4, 5, 10p, 11, and 12 (DELWP 2015)

A PVA model was developed for the species using existing empirical data and elicited expert judgement to investigate the relative merits of alternative management options for biomass control. Fundamental information regarding ecology of the species and the estimation of vital rates was derived from Reynolds (2013). Expert judgement was used to parameterize the model where empirical data was unavailable.

2.2 Methods

2.2.1 Conceptual models

Two conceptual models of the population dynamics of SRF were developed in collaboration with species experts. The first conceptual model outlines the life cycle of SRF, including important life stages, the processes and threats that influence plants survival and reproduction, and how management actions can influence threats (Figure 2.1). The species can be considered in four main stages: seedbank, recruits, juveniles and adults. Seeds can remain in a soil seedbank for several years. When seeds germinate, they become non-reproductive recruits for one year. They can then move to the juvenile stage. Some juvenile plants can reproduce but at much lower rates than adult plants. Individuals can remain as juveniles until approximately five years of age before moving onto the adult stage, where reproduction and survival is higher, and they can remain as adults for approximately 100 years.

The main threats to the SRF within the MSA management area are competitive biomass including weed invasion, prolonged drought events, and inappropriate fire regimes. The accumulation of biomass over time is thought to reduce survival of recruits and juveniles and to a lesser extent in adults. While fire can help to reduce biomass and promote germination, it can also increase mortality of all individuals. If fire occurs prior to or during the flowering and seed production season, it removes the reproductive output for that year (Figure 2.1). Prolonged droughts can reduce survival of juveniles so small-scale watering may be beneficial during times of drought. Higher than average autumn rainfall and low biomass are thought to instigate a germination pulse. Grazing is deemed a large threat to the species. However, it is not included in the conceptual model as we assume that grazing is removed from areas where SRF occurs.

A second conceptual model illustrates the timing of biological processes over a year and the impact of the timing of burning on vital rates (Figure 2.2). Flowering generally occurs between April-August. Germination can occur anytime between May until November. Burning March-November can remove the reproductive output for the year. It can also increase mortality of younger plants. Burning over the summer months has minimal increases in mortality of established plants (Figure 2.2).
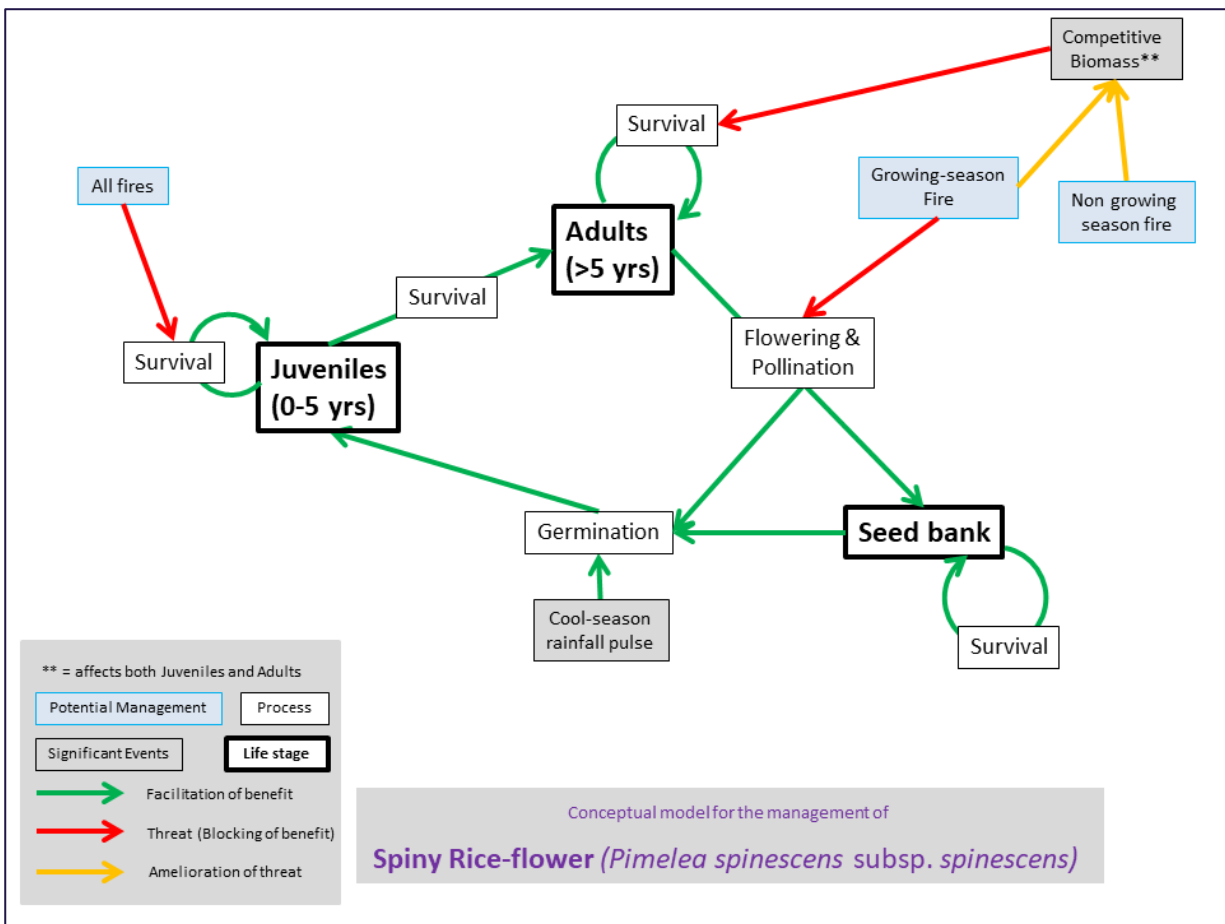
Figure 2.1: Conceptual model of the life cycle of Spiny Rice-flower including how threats and management influence different stages of the life cycle.
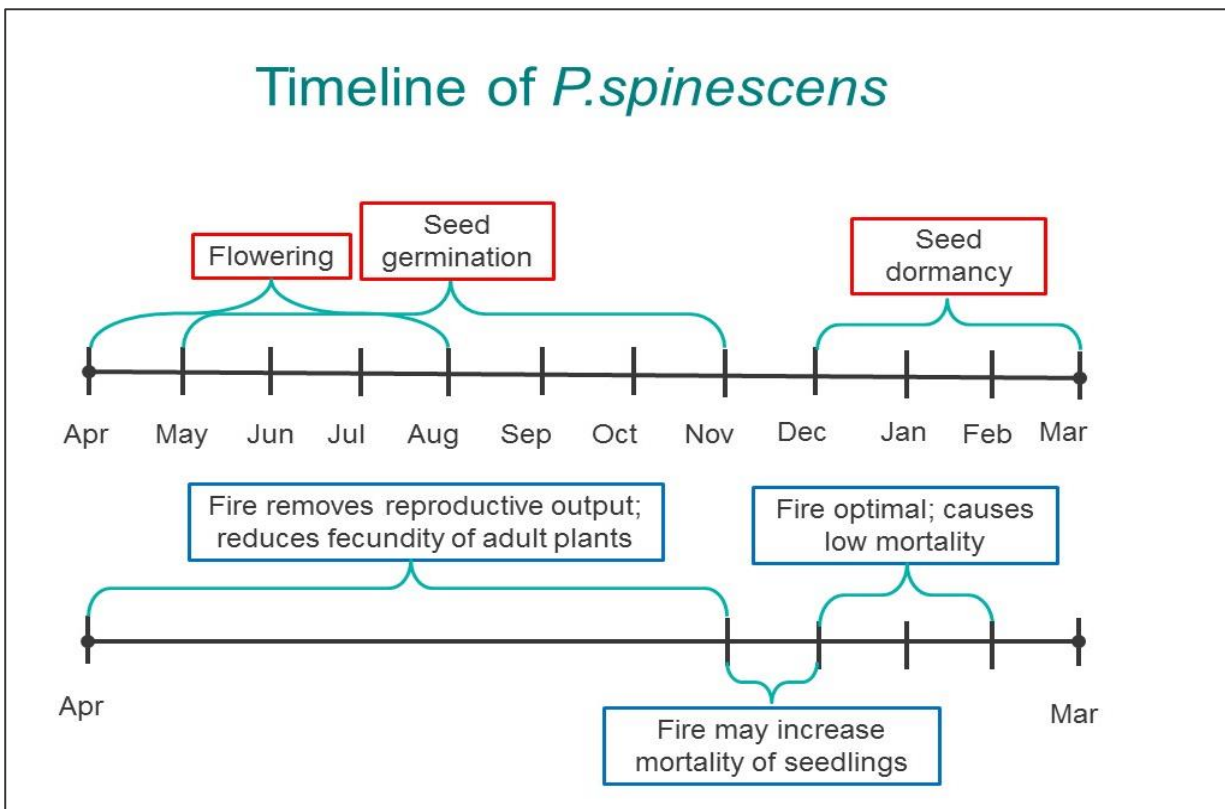
Figure 2.2. Conceptual model of the timing of ecological processes and effects of biomass control in different seasons. Seed dormancy refers to conditions unsuitable for seed germination.

2.2.2 Vital rates

Spiny Rice-flower was modelled using four stage classes; a seed bank, recruits, juveniles, and an adult stage. Vital rates were estimated from empirical data, predominantly from Reynolds (2013). For model parameter estimates lacking empirical data to estimate, expert judgements were elicited using a formal structured process. Vital rates were summarised in Table 2.1.

Fecundity estimates are the number of viable seeds per reproductive individual per year. This was calculated based on estimates of the proportion of female plants flowering per year, the sex ratio, the average number of stems per juvenile and adult plants respectively, the number of seeds produced per stem and the proportion of viable seed. Seeds in the soil seed bank are thought to be viable for up to 6 years based on expert judgements. Germinations rates were derived from estimates of number of germinants per female flowering plants (Reynolds 2013) and a model that estimated the average proportion of seeds that germinate for different population sizes of SRF.

Recruit and adult survival rates were available from Reynolds (2013) and were 0.145 and 0.99 respectively. Juvenile survival was estimated from expert judgements (i.e. 0.74). These vital rates imply a growth rate of 1.02, suggesting a slighting increasing population in the absences of threats. This was assumed to be the growth rate under low biomass conditions, and in the absence of an above average rainfall germination pulse.

Table 2.1: Vital rates for Spiny Rice-flower used in the PVA model

Vital rate	Value	Description
Juvenile fecundity	6.31	Annual number of viable seeds per plant
Adult fecundity	78.86	Annual number of viable seeds per plant
Seedbank survival	0.162	Annual proportion of seeds surviving in the seedbank each year
Seedbank to Recruit	0.0068	Annual proportion of seeds germinating each year and surviving to become new recruits
Recruit to Juvenile	0.145	Annual proportion of recruits surviving to become juveniles
Juvenile survival	0.6	Annual proportion juveniles surviving to the next year
Juvenile to Adult	0.14	Annual proportion of juveniles becoming adults
Adult survival	0.99	Annual proportion of adults surviving to the next year

2.2.3 Stochasticity

Both environmental and demographic stochasticity are included in the model. Environmental stochasticity is modelled by drawing random numbers from lognormal distribution for vital rates with means associated with the values in Table 2.1. Standard deviations use default values of 20% for fecundity and 10% for survival except for adult survival that was set at 1% to avoid stochastic adult survival rates greater than 1.0. Defaults for standard deviations were used due to no empirical or expert estimates. The influence of this assumption was tested in the sensitivity analysis.

Germination pulses due to above average rainfall were modelled as a separate stochastic event. Background germination rates are low (Table 2.1). Above average rainfall was estimated to occur with a probability of 0.125 based on rainfall data from Truganina Cemetery using recent rainfall data (1948-2015) (BoM 2016). This equates to a germination pulse occurring on average every 8 years. A germination pulse was modelled as a probabilistic event, that when it occurs, the germination rates for that year are increased by four times the background rate. This estimate was based on expert judgements.

2.2.4 Biomass Accumulation

Biomass accumulation of native and exotic grassland species is believed to be a major threat to SRF. Under increased biomass, the survival of all plants is reduced as they are outcompeted for light, space and root competition (Reynolds 2013). Biomass accumulation was modelled as a temporal trend in survival and germination rates linked to time since biomass reduction (Figure 2.3). Survival rates under high biomass conditions were elicited from experts and a linear relationship between low and high biomass survival rates was assumed (Figure 2.3). Non-linear relationships were also investigated as part of the sensitivity analysis. Germination rates also decline as a function of time since fire and were derived from estimates in Reynolds (2013). The time since biomass control where biomass was considered high, was inferred from available data collected at Truganina Cemetery, Mount Cottrell and Kalkallo Common (DELWP 2015).
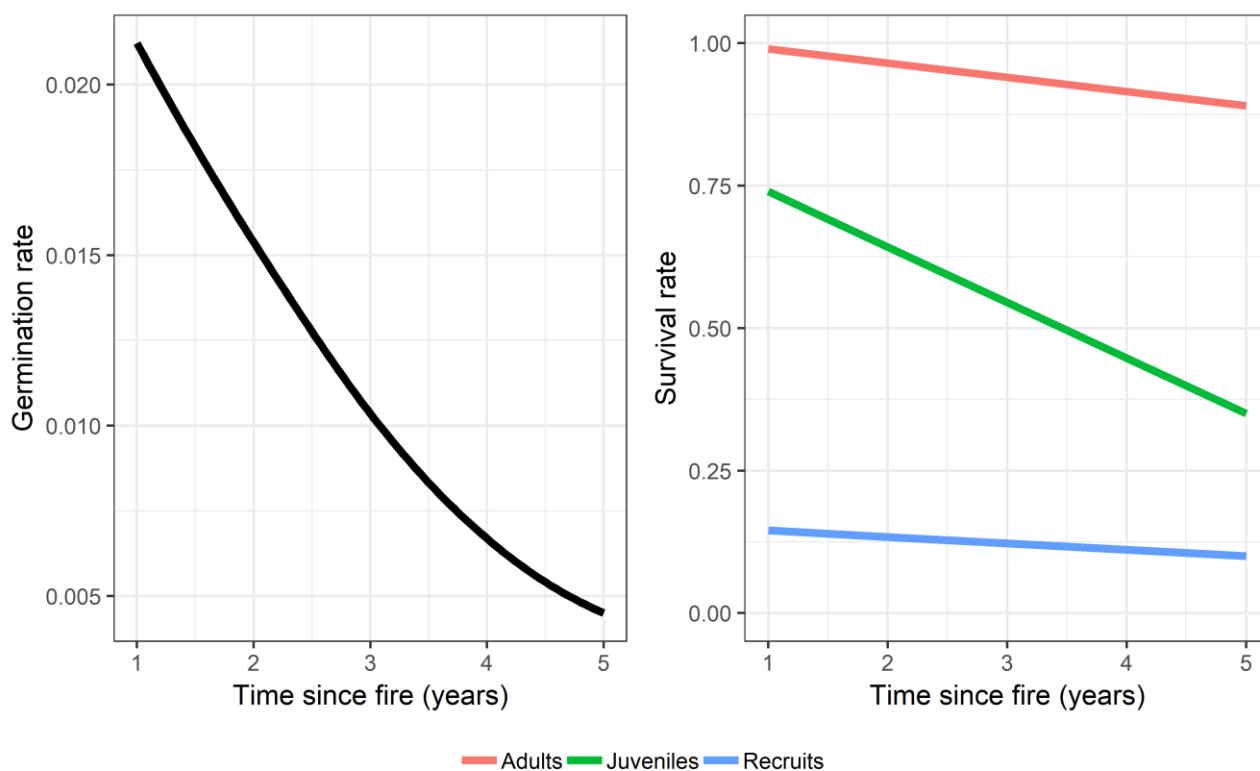
Figure 2.3. Relationship between germination and survival of adults, juveniles and recruits and time since fire (in years).

2.2.5 Additional model specifications

Density dependence: The population is regulated through the relationships between vital rates and biomass accumulation (reduced capacity to grow), and controlled burning (increased capacity to grow). For model convenience, we used a ceiling model where populations can grow exponentially according to the vital rates until they reach a ceiling. Carrying capacities were set high but realistic to avoid any iterations producing unrealistic exponential growth.

Initial abundance: The initial population size was set at approximately 675 plants, to reflect current numbers at Truganina Cemetery and a stable stage distribution was assumed.

The model was run over 50 years for 1000 iterations. The model scenarios were compared using expected minimum abundance (EMA).

2.2.6 Management scenarios

The five standard fire management scenarios for all grassland species were investigated (Table 2.2). While fire reduces biomass and promotes germination, there is an immediate mortality of plants from the fire. When a fire occurs, experts estimate the immediate mortality of recruits, juveniles and adults is 76%, 35%, and 9% respectively. If fire occurs in Autumn, the reproductive output for that year is lost. Due to model constraints, we were not able to model the different effects of fire occurring in different seasons explicitly within the same

model scenario. For model scenarios that included both Autumn and Spring fires, the effect on vital rates was adjusted to account for the different proportions of spring and autumn fires occurring across the 50-year time frame (Table 1.2). Watering during drought was not part of the suite of management actions explored but inferences about watering during drought can be made via the sensitivity analysis results. We assume SRF will be protected from grazing within the MSA management area as the species is sensitive to grazing pressure. Thus, we have not explored the effects of grazing in this study.

Table 2.2: Management scenarios for the Spiny Rice-flower

Scenario number	Description	Fire interval	Fire season
1	No management	No fire management	-
2	Medium fire interval, Autumn	Medium	Autumn
3	Long fire interval, Autumn/Spring	Long	Autumn/Spring
4	Short fire interval, Autumn/Spring	Short	Autumn/Spring
5	Medium fire interval, Spring	Medium	Spring

2.2.7   Sensitivity analysis

The deterministic sensitivity analysis focused on the elasticities of the stage matrix, i.e. the proportional contribution of each of the parameter estimates toward the growth rate implied by the stage matrix. The stochastic sensitivity analysis involved adjusting each of the vital rates and standard deviations of a stage matrix by ±10% to investigate if the stochastic model was sensitive to any these parameters. All survival rates were adjusted together (i.e. recruits, juveniles and adults) and similarly for fecundities.

The scenario sensitivity analysis evaluated two key assumptions in the model to determine whether the rank order of fire management scenarios change compared the base scenarios. The temporal trend in survival rates due to biomass accumulation was assumed to be a linear relationship (Figure 2.3). Two non-linear functions were investigated, a convex shaped curve and a concave shaped curve (Figure 2.4). The convex and concave functional forms were applied to all survival rates in model. Base models were compared to models where fecundity remained constant through time. The most sensitive parameter from the stochastic sensitivity was also investigated to determine whether it influenced the rank order of fire management actions.
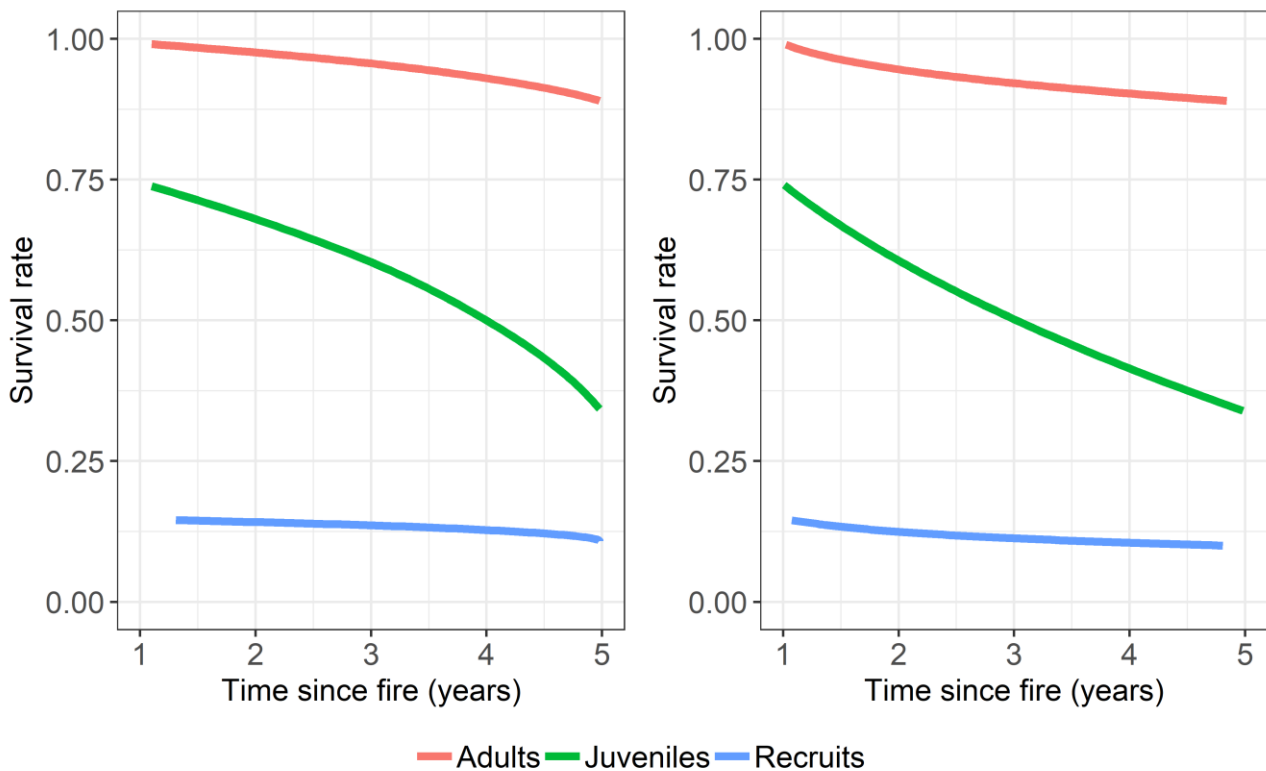
Figure 2.4: Example of different functional forms for trend in survival given TSBC used in the scenario sensitivity analysis.

2.3   Results

2.3.1   Management actions

Results suggest biomass control with short fire intervals is the best management action to support persistence of Spiny Rice-flower (Figure 2.5). Short fire intervals refer to burning every other year. The season of burning is of less importance than the frequency. However, Spring burns seem better than Autumn burns, as the model assumes the reproductive output is lost when burning occurs in Autumn. No fire in the system can be detrimental to the species persistence, the management scenario that didn't include fire as a biomass control (scenario #1 in Figure 2.5) return the lowest EMA amongst all the suites of management actions. When biomass accumulation is not controlled by fire, EMA is reduced by 93% compared to the best EMA achieved by short interval burn scenario. Watering was not part of the management suite in the model but inferences about watering during drought can be made via the sensitivity analysis results. These results are consistent with current recommendations from the Recovery team (Reynolds et al 2017)
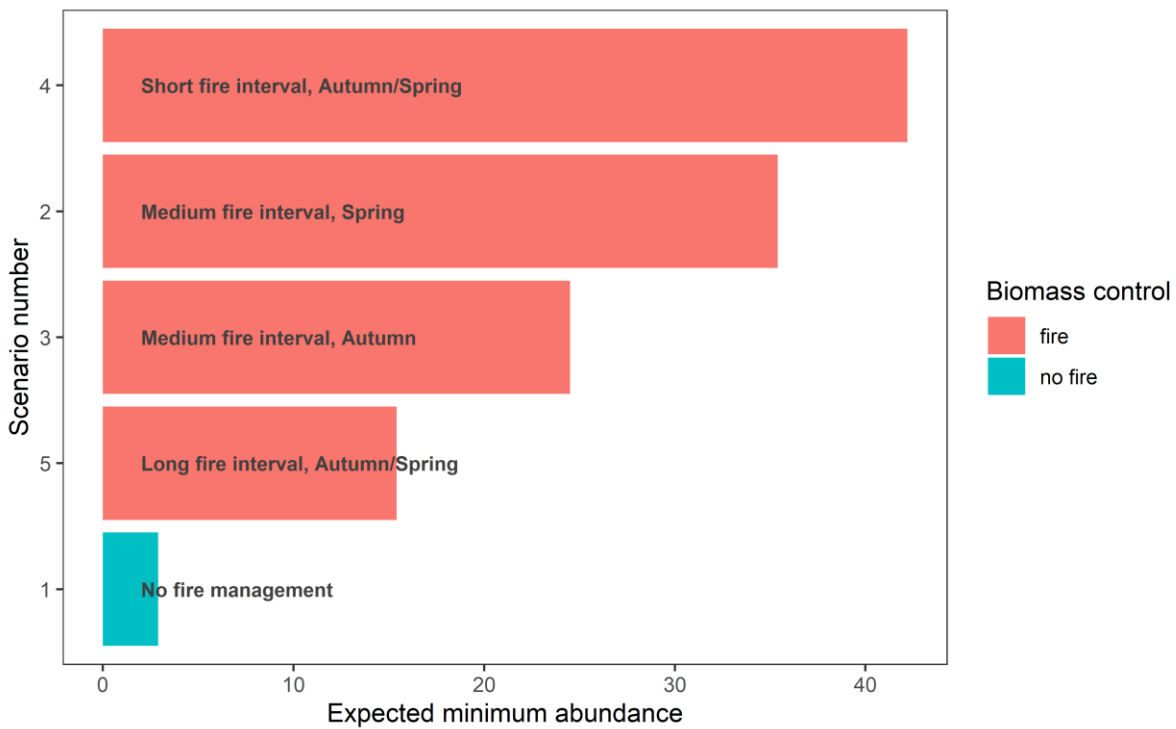
Figure 2.5: The result of PVA for Spiny Rice-flower – ranking of management actions and subsequent EMA for each scenario.

2.3.2 Sensitivity Analysis

The deterministic sensitivity analysis suggests that adult survival contributes to the growth rate the most. The proportional contribution of adult survival to the growth rate is >80%, with the other vital rates providing minimal contributions (<5% each) (Figure 2.6). The sensitivity of survival was reconfirmed by the results of the stochastic sensitivity analysis where ±10% changes in survival rates resulted in percent change in EMA of >60% and ~200% respectively (Figure 2.7). The sensitivity analysis reveals that if survival rates drop dramatically, as they may during drought conditions, it will have a detrimental effect on the population of SRF. This suggests that local scale watering during the growing season, to prevent increases in mortality due to dry conditions, would be beneficial.

Scenario sensitivity analysis revealed the ranks of preferred fire management scenarios for the species are robust to assumptions regarding the functional form (linear) of the temporal trend in survival, at least based on the alternative functions tested. The rank order of the management scenarios was identical irrespective of the functional form used (Table 2.3). While 10% reduction in survival had large changes in absolute values of EMA, the rank order of management actions did not change from base model scenarios.

Unofficial
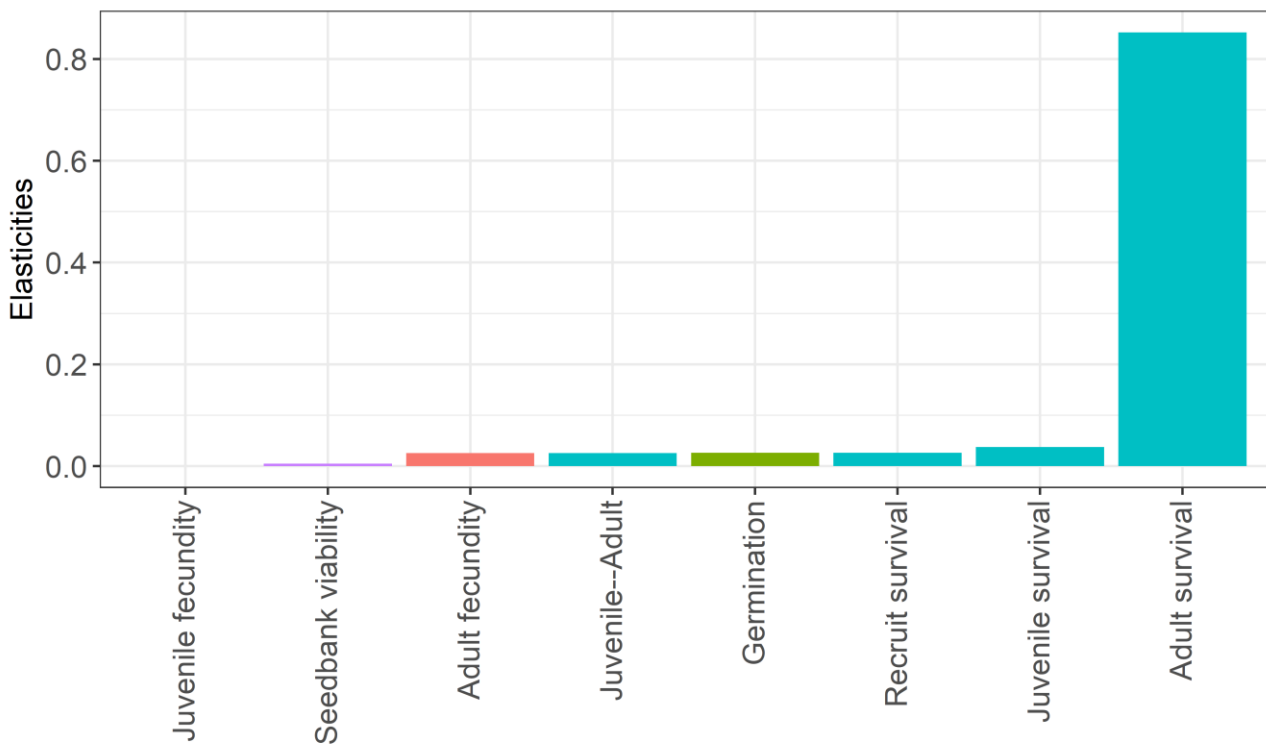
Figure 2.6: Elasticities of the stage matrix for the Spiny Rice-flower.
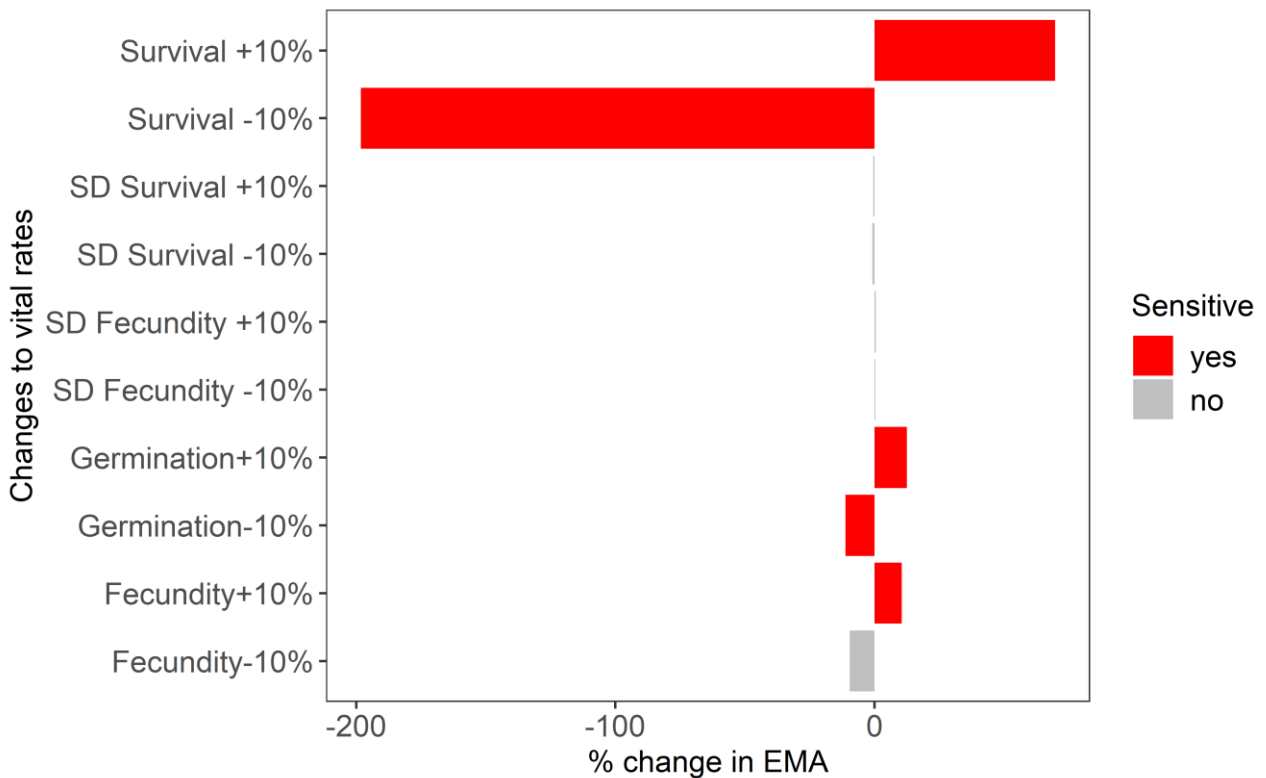
Figure 2.7: Stochastic sensitivity analysis of mean vital rates and standard deviations of the Spiny Rice-flower. Changes in EMA greater than 10% (greater than input change) are coloured in red.

Table 2.3: Scenario sensitivity analysis for Spiny Rice-flower. EMAs and rank order of management actions (in parentheses). Rank correlations are calculated with base models.

Scenario	Base model	When initial abundance =100	Convex shaped	Concave shaped	When survival is reduced by 10%
	EMA (Rank)	EMA (Rank)	EMA (Rank)	EMA (Rank)	EMA (Rank)
Short fire interval, Autumn/Spring	42.2 (1)	5.1 (1)	72.2 (1)	29.4 (1)	0.19 (1)
Medium fire interval, Spring	35.4 (2)	4.3 (2)	67.6 (2)	25.5 (2)	0.11 (2)
Medium fire interval, Autumn	24.5 (3)	3.0 (3)	49.0 (3)	18.8 (3)	0.10 (3)
Long fire interval, Autumn/Spring	15.4 (4)	1.7 (4)	37.4 (4)	15.6 (4)	0.06 (4)
No management	2.9 (5)	0.4 (5)	15.1 (5)	8.3 (5)	0.01 (5)
Correlation coefficient		**1.0**	**1.0**	**1.0**	**1.0**

2.4   Discussion

Spiny Rice-flower is a critically endangered species requiring specialised management within the MSA management area for its ongoing persistence. This study brings together current knowledge of the life history and ecology of the species into a quantitative model so the relative benefits of alternative management actions can be explored in a systematic way. Four management actions focusing of different fire seasons and frequencies were explored and compared to a no management scenario. Burning at short intervals every second year is the preferred management for the species. While burning frequently increases mortality due to some plants being killed by fire, this mortality is less than the mortality and reduced germination if competitive biomass remains unmanaged. Frequency of burning is more important than season, although it is preferable to avoid burning prior to or during the reproductive phase of the species. While specific scenarios for watering during drought were not explored, sensitivity analysis suggest that a reduction in survival can lead to large increases in extinction risk. Survival is likely to be reduced during drought years, so watering plants will help safeguard populations during drought years. This may become more important under a drying climate.

The large influence of adult survival on model behaviour suggests investigating an alternative model structure may be warranted. For instance, including multiple adult stages may provide some extra nuance within the adult stage that may be important for the population dynamics and subsequent management. There is a trade-off, as this added complexity will require more information to parameterise the model and it is difficult to accurately age individual plants.

The sensitivity analysis revealed the model is most sensitive to survival rates, particularly adult survival. Scenario sensitivity analysis revealed the management actions explored were robust to assumptions regarding the functional form of the trend in survival and germination rates given time since biomass reduction. Ranks were also robust to changes in survival rates of the stage matrix. These results are encouraging as this suggests that the influence of these assumptions has little effect on preferred management actions. More sensitivity analysis is required to fully explore the effect of assumptions in the

Unofficial

model. In particular, sensitivity associated with assumptions of germination pulse, probability of above average rainfall, and shifts in temporal trends of survival and germination. These sensitivities should be explored in the next iteration of the model.

It is thought that SRF requires above average rainfall in Autumn to stimulate a germination pulse and follow-up rain events to support survival of the rain-stimulated cohort. For all model scenarios we used a probability of 0.125 which equates to above average rainfall approximately every eight years. If the interval between above average rainfall increases, or follow up rain events decline, as is likely with climate change, this could result in very little recruitment. In this case localised watering, to simulate a rainfall pulse and increase survival may be needed as a management action to safeguard populations under climate change. Specific climate change scenarios and alternative adaptation strategies could be explored in future iterations of the model.

3 Button Wrinklewort (*Rutidosis leptorhynchoides*)

3.1 Background

Button Wrinklewort (BWW) is a multi-stemmed perennial forb native to the temperate grasslands and grassy woodlands of south-eastern Australia. BWW was once quite abundant, however herbarium records suggest that there has been a significant reduction in the number and size of populations since 1874 (Young and Murray 2000). The species is pollinated by a range of flying insects and is known to be primarily self-incompatible (Young et al. 1999). Seed dispersal distances are generally less than 0.5 m (Morgan 1999a). Flowers are produced from November to March, with seedlings emerging in late autumn, generally after the first major rains of the season. There is no long-term soil-stored seed bank (Morgan 1999b). Seedlings are thought to be the most sensitive life-cycle stage; however, once established as an adult, BWW plants are robust and resilient to fire and drought but not grazing or competitive exclusion (Morgan 1997, 1999a). In grasslands near Melbourne competitive exclusion by dense stands of the dominant grass (*Themeda triandra*) is the most significant factor restricting germination and survival (Scarlett and Parsons 1990a, b, Morgan 1997). Current management recommendations include regular burning to promote recruitment and survival of BWW by increasing the intra-patch spaces. Inter-fire interval of two to four years is recommended in order to capitalise on occasional rainfall events, longer inter-fire interval reduces recruitment and increases seedling mortality (Morgan 1997).

3.2 Methods

3.2.1 Conceptual model

The conceptual model below (Figure 3.1) outlines Button Wrinklewort's ecology and key life history stages, the processes and drivers that influence the plant's survival at each stage, and the management actions that can be used to ameliorate these threats and increase plant survival and fecundity. This simplified conceptual model informed the development of the management population models for the species.

The timeline of important reproductive events and responses to fire season (Figure 3.2) provides context for the way management can potentially ameliorate the threats of biomass accumulation. The amelioration of such threats should improve fecundity and survival of BWW; however, it is important to note that the timing of management can have a negative effect on the fecundity and survival of Button Wrinklewort (e.g. if fire implemented after seed germination in Autumn).
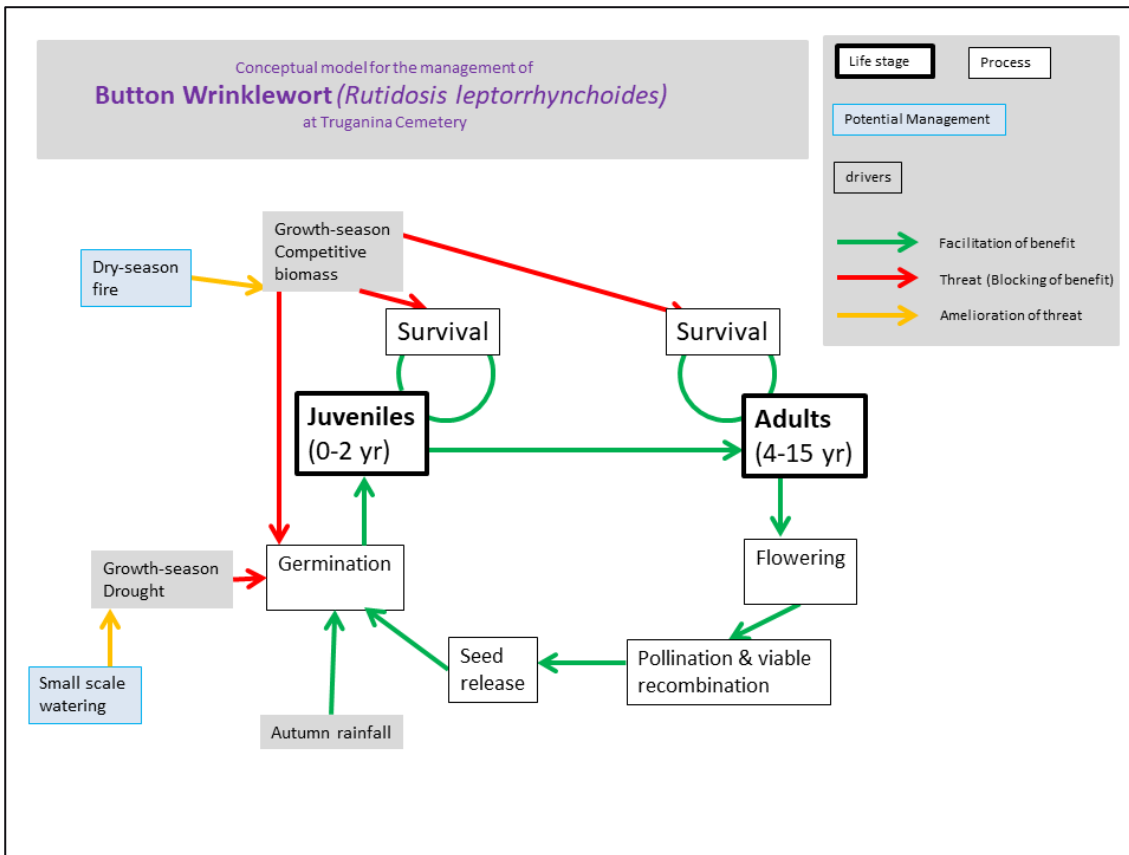
Figure 3.1: Conceptual model of the life cycle of Button Wrinklewort including how threats and management influence different stages of the life cycle.
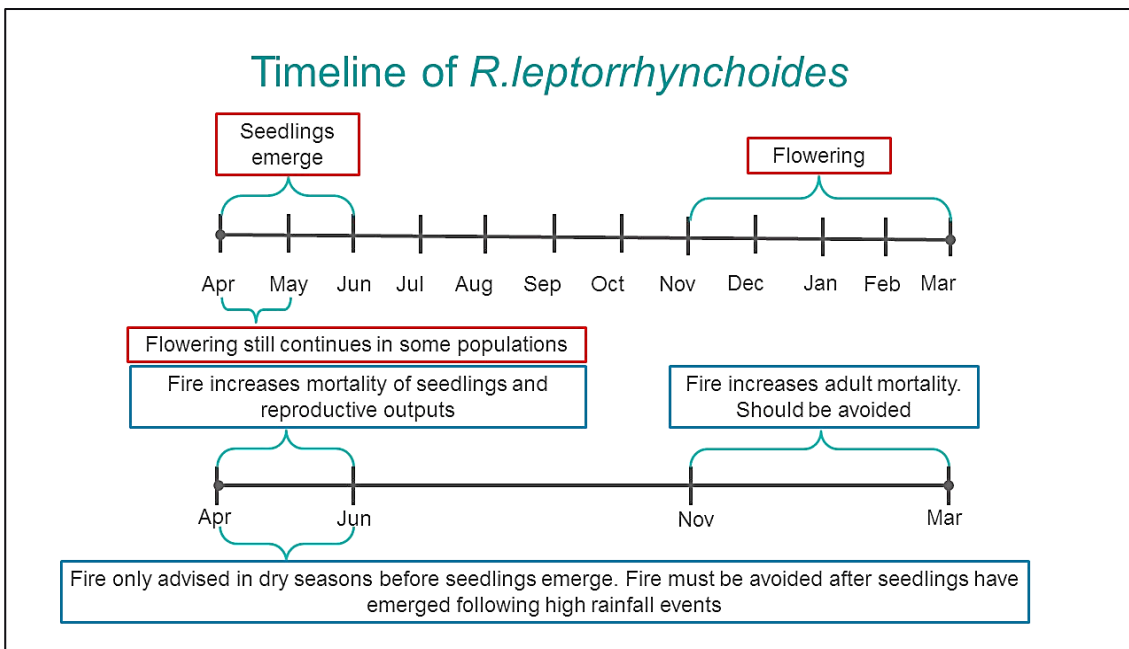
Figure 3.2: Conceptual model of the timing of ecological processes and effects of biomass control in different seasons BWW.

3.2.2 Vital rates

Button Wrinklewort was modelled as an age/stage matrix population model (Caswell 2001) assuming a pre-reproductive census. Juveniles were modelled in two age classes (i.e. Juvenile 1 and Juvenile 2) representing individuals aged one and two years old. When individual plants are almost three years old, they move to the adult stages. Adult plants are modelled as two stages, adults and senescent adults. Once a plant becomes an adult, it has the potential to reproduce. Individual adult plants remain in the adult stage for approximately 13 years before they become senescent. Senescent plants are individuals that have reduced survival and fecundity compared to adult plants. There is no specific stage for seedlings. Instead, survival of seedlings to one-year old juveniles are subsumed in the fecundity rate. See below.

Survival rates

Parameter estimations were derived from empirical studies and expert elicitation sessions. Survival rates in the base model (grassland biomass accumulates, no biomass control, rainfall pulse 1 in 8 years) were estimated for four stage matrix components; juvenile 1, juvenile 2, adult and senescent (Table 3.1). Survival rates of juvenile and adult stages were estimated to be 0.95 (combination of expert elicitation and empirical studies (Bau unpublished, Young and Pickup 2010) and survival of senescent plants was estimated to be 0.5 based on expert elicitation.

Fecundity

Effective fecundity rate was determined by five components and estimated using published literature (Morgan 1995a, 1999a) and MSA monitoring data. Firstly, the number of inflorescences per plant; percentage of inflorescences aborted per plant, and number of seeds produced per inflorescence were combined to calculate the number of seeds produced per plant. Secondly, the percentage of seedling emerging and the percentage of seedlings surviving the first 61 weeks were used to calculate the number of seedlings surviving the first 61 weeks post-germination per plant. Fecundity rate for adult plants was estimated to be 0.093. This fecundity rate represents the number of one-year old juvenile plants (i.e Juvenile 1) produced per adult plant that survives to the next census. The fecundity rate for senescent individuals is thought to decease by 30% from that of adult fecundity rate based on expert judgement (Table 3.1).

The growth rate (i.e Lambda λ) implied by these vital rates was λ = 0.9916. This growth assumes low biomass condition and no germination pulse. The effects of biomass accumulation and a germination pulse from above average rainfall are modelled as separate events. Details are available in the following sections.

Table 3.1: Vital rates of Button Wrinklewort in the absence of germination pulse and under low biomass conditions

Parameter	Value	Description
Adult fecundity	0.093	Number of juveniles produced per adult plant per year
Senescent fecundity	0.065	Number of juveniles produced per senescent plant per year
Juv 1 to Juv 2	0.95	Annual proportion of 1st year juveniles surviving to their 2nd year
Juv 2 to Adult	0.95	Annual proportion of 2nd year juveniles surviving to adulthood
Adult survival	0.9	Annual proportion adults surviving to the next year
Adult to Senescent	0.05	Annual proportion of adults becoming senescent
Senescent survival	0.5	Annual proportion of senescent individuals surviving to the next year

3.2.3 Stochasticity

Demographic and environmental stochasticity were included in the model. Environmental stochasticity is modelled using a lognormal distribution for vital rates with means associated with the values in the stage matrix and standard deviations based that use default values of 20% for fecundity and 10% for survival rates.

Defaults for standard deviations were used due to no empirical or expert estimates. The influence of this assumption was tested in the sensitivity analysis.

Germination pulse

A pulse of germination occurs with above average autumn rainfall. The average autumn rainfall between 1942-2016 is 130 mm. Above average Autumn rainfall was estimated to occur with a probability of 0.125 based on BOM data for Laverton. This equates to a germination pulse occurring on average every 8 years. A germination pulse was modelled as a probabilistic event, that when it occurs, the germination rates for that year are calculated as: germination (pulse year) / germination (average year).

3.2.4 Biomass accumulation

Biomass accumulation can have adverse effects on BWW population processes if not managed appropriately. Direct effects of biomass accumulation in the PVA were considered on two independent parameters; fecundity and survival rates.

Effects on survival rate

Although expert elicitation indicated that the survival rate of Button Wrinklewort declines as time since biomass control increases the shape of the function between low (1 year since biomass control) and high biomass (5+ years since biomass control) was not directly elicited. Subsequent consultation with grassland experts suggested that the function was non-linear, and resembled an exponential decay function where the decline to survival rate accelerates after three years since biomass control (Figure 3.3). The elicited values represent relative change.

Effects on fecundity

Estimations based on empirical data (Morgan 1997) suggest that fecundity rates and time since biomass control have an inverse linear relationship where fecundity rate declines to 0 in high biomass conditions for reproductive life stages (5+ years since biomass control). Juvenile plants are not reproductively active, so fecundity rates remain unchanged over the time since biomass control gradient (Figure 3.3).
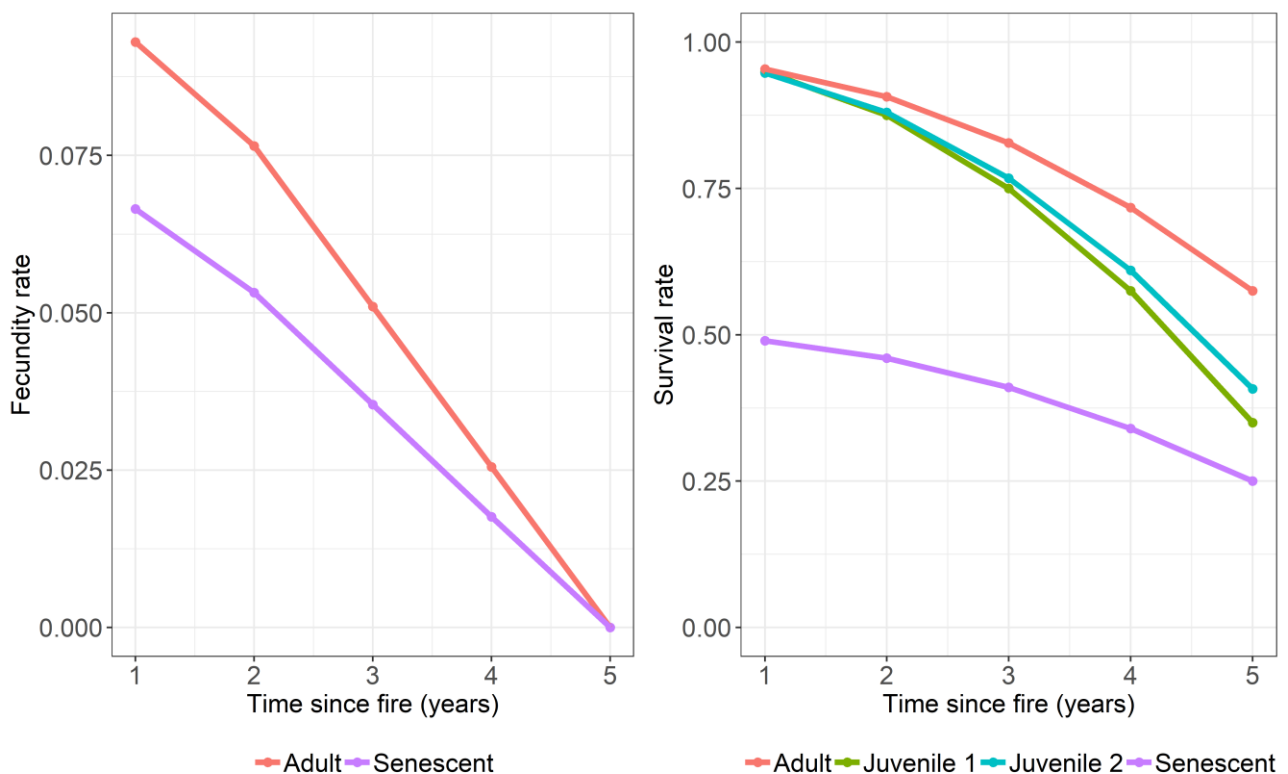
Figure 3.3: Relationship between fecundity and survival rates of four life stages and time since biomass control.

3.2.5 Additional model specification

Density dependence: In the model, population dynamics are regulated through the relationship between vital rates and biomass accumulation (reduced capacity to grow), germination pulses due to higher than average rainfall events, and controlled burning (increased capacity to grow). For model convenience, we used a ceiling model where populations can grow exponentially according to the vital rates until they reach a ceiling. Carrying capacities were set high (i.e. at 5000 individuals) to avoid any iterations producing unrealistic exponential growth.

The accumulated average residency across life stages in the PVA is 14 years which is equivalent to estimated longevity (Scarlett and Parsons 1990b).

Initial abundances: Based on 2016 census at Truganina cemetery, the initial population size was set at 600. Initial abundances were assumed to be at a stable age distribution. The model was simulated 1000 times over a 50-year time frame into the future (i.e. 1 time step = 1 year). The model scenarios were compared using expected minimum abundance (EMA), the minimum abundance in a 50-year time span averaged across all 1000 simulated trajectories of the model.

3.2.6 Management scenarios

Biomass control - Fire

Planned fires in grasslands are often applied in either spring or autumn. BWW plants can either be advantaged or disadvantaged in the short-term depending on the season of the burning, and how the timing of the burn relates to autumn rainfall break (see Figure 3.2 above).

- Fire prior to the autumn rainfall break is deemed best management practice as both adults and germinants are yet to resprout or germinate, and will only experience very minor disruptions to survival and fecundity.

- Fire after the autumn rainfall break can cause 100% mortality of all recently germinated seedlings (the entire reproductive output from the previous year as there is no persistent seedbank), while adults will only experience very minor disruptions to survival and fecundity the following year. As this fire regime is clearly undesirable for the long-term persistence of BWW it has not been considered as part of the population viability analysis in this report.

- Fire in spring can cause extensive reductions in reproductive output (fecundity ~0) in the following year due to the burning of immature reproductive structures (buds, flowers, seeds) prior to seed release in summer, while adult plants will only experience very minor disruptions to survival.

Although fire ideally occurs in autumn prior to the rainfall break it may not be possible when biomass control is required due to the prevailing conditions and resources (e.g. drought, wildfire). Therefore, spring fires have been incorporated into the fire season scenarios to allow biomass control to continue in a suboptimal season as biomass accumulation is deemed a greater threat to Button Wrinklewort population persistence compared with spring fires (Table 3.2).

The season in which a fire occurs effects fecundity rates differently. Specifically, spring fires reduce reproductive output in the year it occurs, while fires that occur in Autumn have only a minor impact on fecundity rates. Due to model constraints, we were not able to model the different effects of fire occurring in different seasons explicitly within the same model scenario. For model scenarios that included both Autumn and Spring fires, the effect on fecundity rates was adjusted to account for the different proportions of spring and autumn fires occurring across the 50-year time frame.

When fires occur in spring, we assume the reproductive output for that year is lost. For spring fires, fecundity was reduced to 0.0 for the year the fire occurs. Fecundity was reduced to: 0.75 in fire scenarios where spring fires occur 1 in every 4 fires (Proportion of spring fires = 0.25), and 0.5 in fire scenarios where spring fires occur half the time (Proportion of spring fires = 0.5). Where only autumn fires are implemented the fecundity rate was not adjusted (Proportion of spring fires = 0).

Mortality of established plants due to fire is the same irrespective of season. Juvenile mortality is increased by 20% adult mortality is increased by 10% in the event of a fire. These estimates were based on expert judgement.

Watering

We assume the effects of watering are equivalent to the effects given a rainfall pulse. To simulate watering, the probability of a rainfall pulse (i.e. 1 in 8 years or Pr(Pulse) = 0.125) was increased to 1 in 6 years (or Pr(Pulse + Watering) = 0.167). The simulated watering event was applied to all fire season scenarios to

determine which scenario had the greatest benefit for Button Wrinklewort population persistence over 50 years (Table 3.2).

Table 3.2: Summary of management scenarios considered in each model.

Scenario number	Description	Fire interval	Fire season	Watering regime
A1	No management	No fire management	-	-
A2	Short fire interval Spring/Autumn, no watering	Short	Spring/Autumn	No watering
A3	Medium fire interval Autumn, no watering	Medium	Autumn	No watering
A4	Medium fire interval Spring, no watering	Medium	Spring	No watering
A5	Long fire interval Spring/Autumn, no watering	Long	Spring/Autumn	No watering
W1	No fire management, watering	No fire management	-	Watering
W2	Short fire interval Spring/Autumn, watering	Short	Spring/Autumn	Watering
W3	Medium fire interval Autumn; watering	Medium	Autumn	Watering
W4	Medium fire interval Spring; watering	Medium	Spring	Watering
W5	Long fire interval Autumn/Spring, watering	Long	Spring/Autumn	Watering

3.2.7 Sensitivity analysis

Three sensitivity analysis approaches were employed to examine the effect of changes in vital rates on model outcomes and the ranking of management actions. The deterministic sensitivity analysis focused on the elasticities of the stage matrix, i.e. the proportional contribution of each of the parameter estimates toward the growth rate implied by the stage matrix. Stochastic sensitivity analysis involved changing the survival rates, fecundity and the standard deviation of survival and fecundity by ±10%. The scenario sensitivity analysis explored the options of linear and nonlinear relationships (exponential) between fecundity and biomass control, and how these scenarios effect the ranking of the management actions (Figure 3.4).
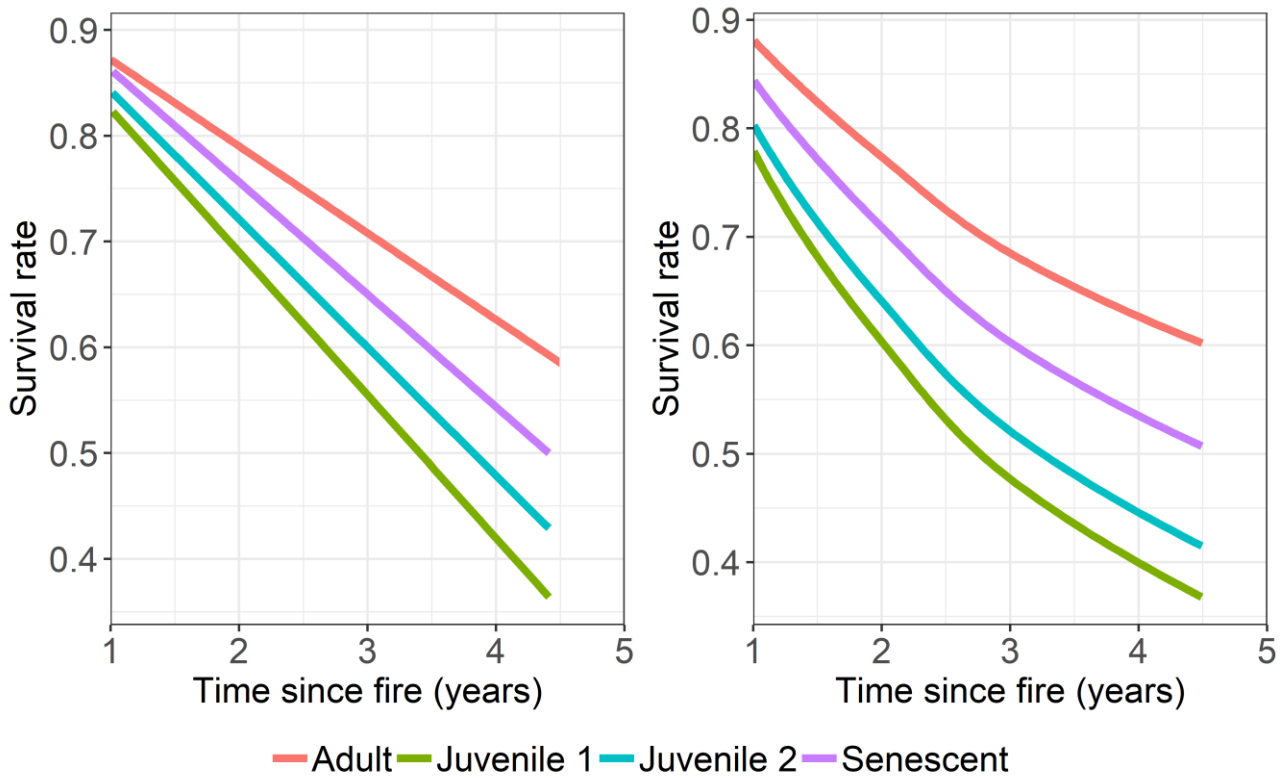
Figure 3.4: Example of two different functional forms for trend in survival given time since fire (i.e biomass control) used in the scenario sensitivity analysis.

3.3 Results

3.3.1 Management scenarios

Results suggest short fire intervals in either Autumn or Spring and watering is the preferred management option that promotes persistence of BWW population the most (Figure 3.5). Short fire intervals refer to burning every 2 years. Short fire intervals combined with watering have a much larger positive effect than fire or watering has alone, suggesting that combined management is having a synergistic positive effect on population persistence (Figure 3.5). Watering without managing biomass has little benefit to the population. Fire interval seems to have a larger effect than fire season. However, Autumn burns are preferable to Spring burns as reproductive output is lost when burning occurs in Spring. Some form of management is required to sustain BWW populations. If BWW populations are not managed, they are likely to become locally extinct within the next 50 years.

Unofficial
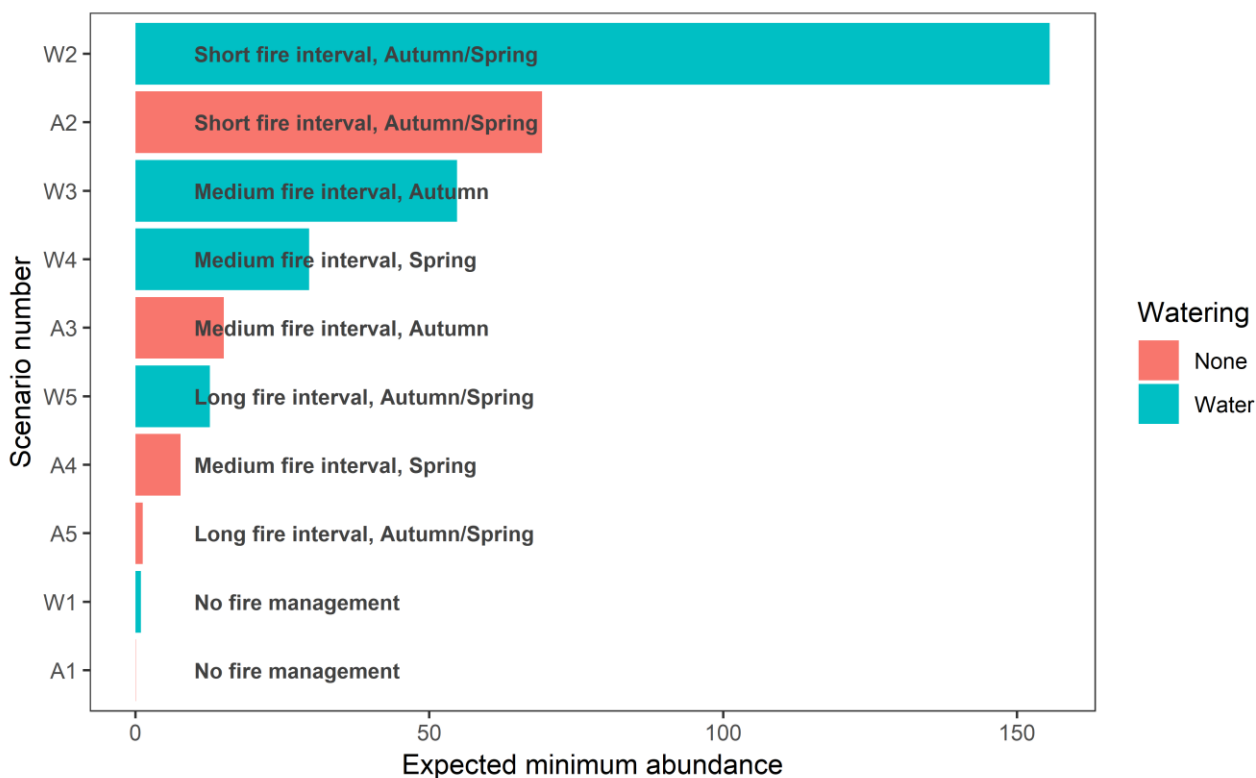
Figure 3.5: The result of PVA for Button Wrinklewort – ranking of management actions and subsequent EMA for each scenario.

3.3.2   Sensitivity analysis

The deterministic sensitivity analysis revealed that adult survival is contributing the largest proportion to the growth rate (i.e. 0.76) with the other vital rates providing small contributions (Figure 3.6).

Stochastic sensitivity analysis revealed the model is sensitive to changes in survival rates and fecundity rates. A 10% decrease in survival rate reduced the expected minimum abundance by more than 150%. A 10% increase in survival resulted in an 80% increase in EMA. The model was only sensitive to decreases in fecundity, with a 10% reduction resulting in >10% change in EMA (Figure 3.7). The model was not sensitivity to any other parameters tested.

The scenario sensitivity revealed that the ranks of management actions are robust to our assumptions regarding the functional form of the trend in survival given time since biomass reduction. Alternative functional forms resulted in rank correlations of 0.98 and 0.99 for the linear and exponential functions respectively. Reduction in survival rates did change the rank order of management actions resulting in a correlation of only 0.48. This was mainly due to the species approaching extinction when survival rates are reduced, making it difficult to differentiate between any benefits gained from the alternative management actions (Table 3.3).

In the scenario sensitivity analysis, when survival rates were reduced by 10%, the ranks of the preferred management actions changed dramatically. Actions 7, representing a short fire interval with watering changed rank from 1 to 8. This suggests the model outcomes with respect to preferred management is not robust to uncertainty in survival rates (Table 3.3).
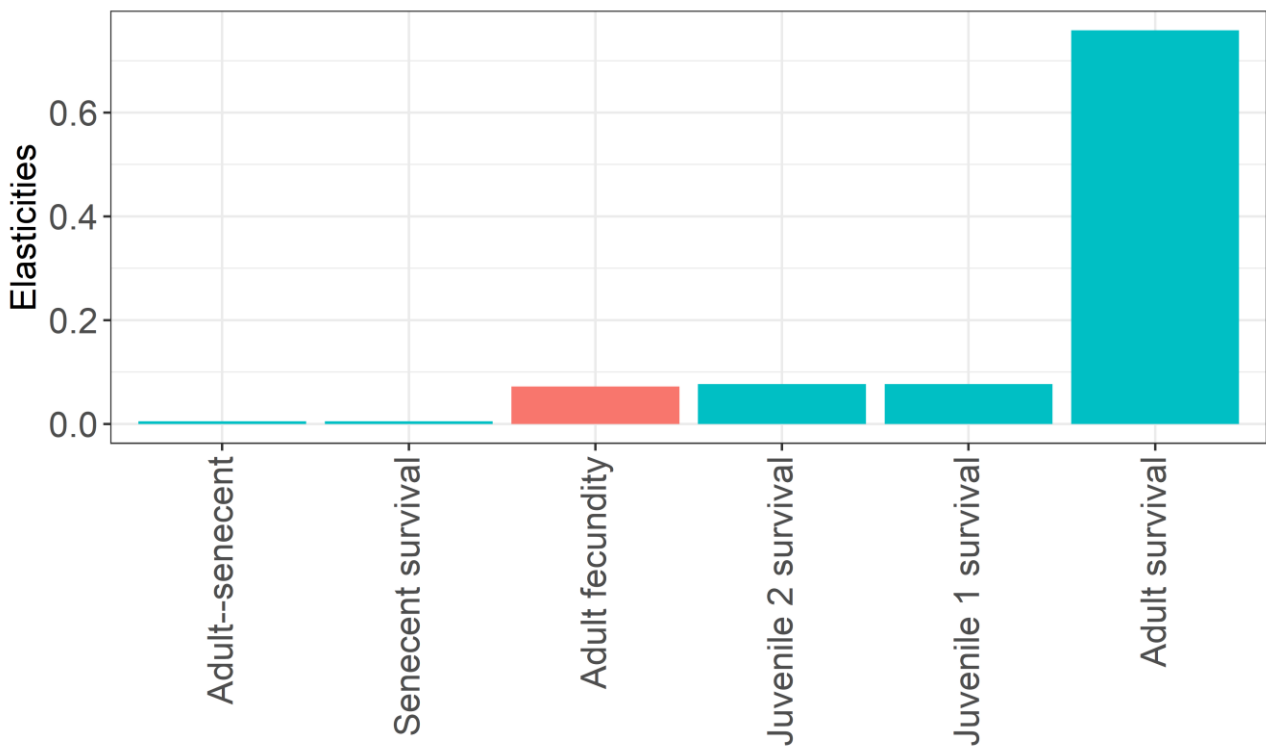
Figure 3.6: Elasticities of the stage matrix for the Button Wrinklewort.
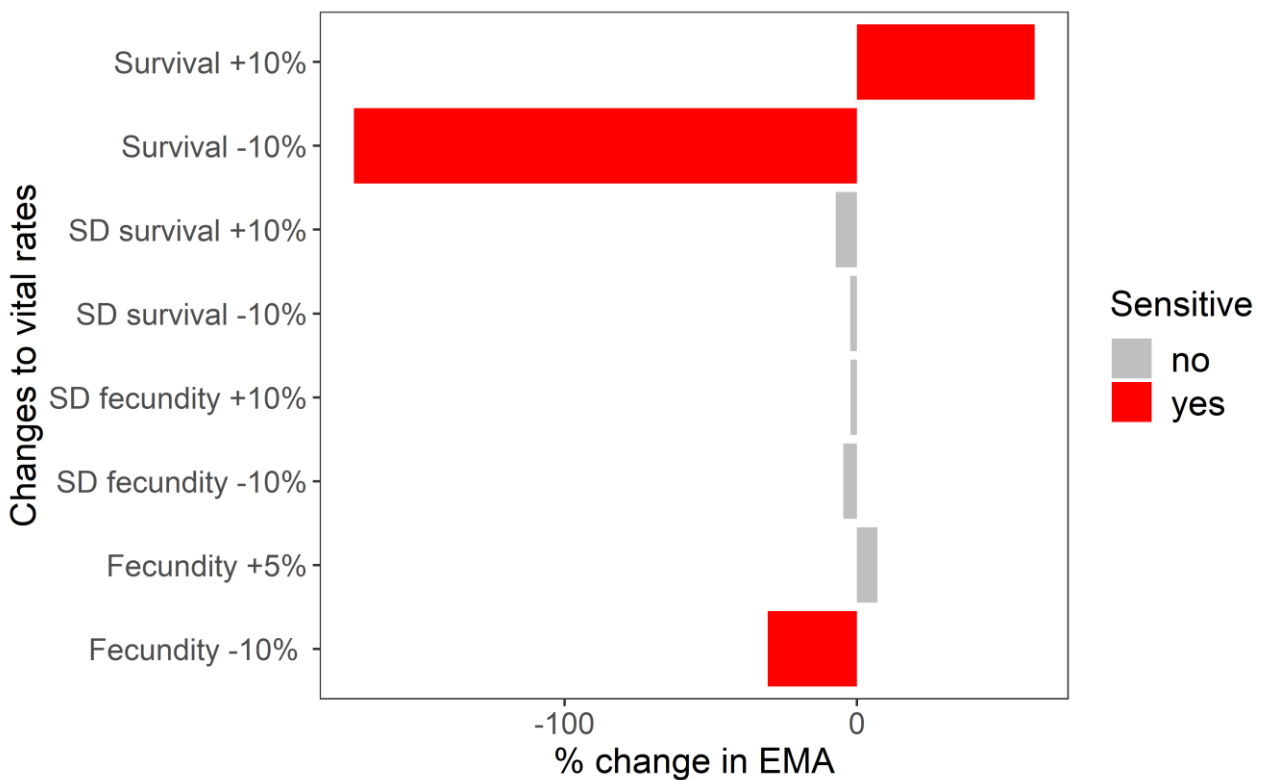
Figure 3.7: Result of stochastic sensitivity analysis on scenario 7 (Short fire interval, Autumn/Spring, no watering). Changes in EMA greater than 10% (greater than input change) are coloured in red.

Unofficial

Table 3.3: Scenario sensitivity analysis for BWW showing ranks of management actions for different model assumptions. Rank correlations are with base models.

Model No.	Management scenario	Base models	Linear function	Exponential function	10% reduction in survival
		EMA (Rank)	EMA (Rank)	EMA (Rank)	EMA (Rank)
1	No management	0.022 (10)	0.002 (10)	0.006 (10)	0 (10)
2	Short fire interval Spring/Autumn, no watering	69.172 (2)	3.462 (3)	18.138 (2)	0.426 (6)
3	Medium fire interval Autumn, no watering	15.006 (5)	0.401 (6)	1.687 (6)	2.7 (3)
4	Medium fire interval Spring, no watering	7.684 (7)	0.266 (7)	0.979 (7)	0.637 (5)
5	Long fire interval Spring/Autumn, no watering	1.224 (8)	0.053 (8)	0.24 (8)	0.175 (7)
6	No fire management, watering	0.896 (9)	0.015 (9)	0.146 (9)	0.027 (9)
7	Short fire interval Spring/Autumn, watering	155.588 (1)	19.203 (1)	54.471 (1)	0.094 (8)
8	Medium fire interval Autumn; watering	54.738 (3)	4.615 (2)	12.224 (3)	13.838 (1)
9	Medium fire interval Spring; watering	29.537 (4)	1.888 (4)	8.351 (4)	4.906 (2)
10	Long fire interval Autumn/Spring, watering	12.632 (6)	1.185 (5)	2.505 (5)	1.495 (4)
	Correlation co-efficient		0.98	0.99	0.48

3.4   Discussion

We developed a population viability analysis model for BWW using existing data supplemented by multiple expert judgments to explore the effectiveness of several management options on the persistence of the species. This will inform the development of a management plan. Results suggest a combined management plan of short fire intervals (every 1-3 years) and watering plants to simulate a rainfall pulse is needed to help safe guard the species from extinction. Without combining all both actions, the effectiveness of any one action is substantially reduced. Furthermore, Autumn fire is preferable to Spring fire.

Currently the population at Truganina cemetery takes refuge in the low-biomass context of the low-biomass halos around trees. Here, the species persists regardless of biomass build up elsewhere at the site. When using the model outcomes to inform specific decision Truganina, it is important to account and factor in these site-specific factors. In addition, before using these results to develop a management plan for the species, consultation with all experts involved in the elicitation will be required to ensure interpretation of model results is correct and recommendations for management are sound.

Sensitivity analysis revealed the model is most sensitive to survival, particularly a reduction adult survival, and moderately sensitive to fecundity. We investigated the influence of reducing survival by 10% on the ranks of management scenarios and found the model is not robust to perturbations in survival rates. This means if actual survival rates are much lower than estimated in the model, the preferred actions for the species may not be well captured in model outcomes. This suggests more investigation into survival rates and how they are modelled is needed. Current monitoring within the MSA program that tracks individual plants from year to year is crucial data for accurately estimating survival rates within the management area. Alternative model structures, such as the inclusion of multiple adult stages, may add extra nuance which could be important for understanding the impacts of management on different life stages. Ongoing data collection on survival of plants will help to ensure, survival estimates are as accurate as possible.

We also investigated two alternative assumptions regarding trends in vital rates in response to biomass control, a linear and an exponential decay function. Neither of these functions significantly impacted the ranks of management actions, although assuming a linear function elevated medium frequency Autumn fire with watering to second, above short frequency fire without watering.

BWW is one of the most well studied threatened species in the MSA area. There are however several areas of research that could improve the model. BWW has a breeding system controlled by single locus self-incomparability. This system is advantageous in large populations but deleterious in small populations as it contributes to small effective population sizes and inbreeding depression (Young and Murray 2000). Genetic rescue is a proposed technique to help alleviate this problem and research is underway. The results of that study will likely contribute to future iterations of this model. Another threat not included here is the cumulative effects of seed collection which may have negative impacts on recruitment. The scale and impact of this remains unknown (J. Morgan, pers. com.) and is worthy of inclusion should more information become available.

BWW is thought to rely on a rainfall pulse in Autumn for recruitment. We modelled the occurrence of this rainfall event to be once in every eight years. However, if under climate change these events become less frequent, then specific actions such as more frequent watering to stimulate germination should be considered.

Unofficial

4 Matted Flax-lily (*Dianella amoena*)

4.1 Background

Matted Flax-lily (*Dianella amoena)* is a small, perennial, rhizomatous lily endemic to Victoria. Plants form clumps up to five meters wide. Flowering occurs from October to April (Carr and Horsfall 1995)). Pollination is effected by native bees, as all *Dianella* species are 'buzz pollinated', where pollen is released from the pollen tube in response to high frequency vibration from the wing beats of native bees (Carr and Horsfall 1995).

Matted Flax-lily (MFL) is currently known to occur in Victoria, where it is distributed from the south-west to the east of the state, and around Canberra. It occurs in grassland and grassy woodland habitats. Much of this habitat has been cleared, and remaining populations of Matted Flax-lily are mostly small and isolated from each other. Current threats include ongoing clearing of habitat and weed invasion (Carter 2010). It is listed as Endangered under the Commonwealth *Environment Protection and Biodiversity Conservation Act* 1999.

The total population of MFL is thought to be approximately 2,500 plants(Carter 2010) recorded at about 120 sites. However, individuals are sometimes difficult to distinguish in the field because of the rhizomatous habit of the species. Therefore, the total number of reproductive individuals may be much less.

4.2 Methods

4.2.1 Conceptual models

A conceptual model was developed that outlines the MFL's ecology and key life history stages, the processes and drivers that influence the plant's survival at each stage, and the management actions that can be used to ameliorate these threats and increase plant survival and fecundity (Figure 4.1). The species has been described in three main stages, seedbank, juveniles (age 0-3) and established adults (age>3). Plants become reproductive above 3 years of age. Seeds are thought to persist in soil more than a year, and hence we incorporated a seedbank stage. This simplified conceptual model informed the development of the management population models for the species.

The main threats to MFL are competitive biomass, prolonged drought, sustained grazing from livestock and inappropriate burning regimes. MFL may also be susceptible to allelopathic effects from exotic grasses (G. Carr, pers. comm.). The accumulations of biomass over time is thought to reduce the survival of all plants but at different rates depending on juvenile or adult life stages (Figure 4.1). Prolonged drought increases mortality of plants.

The main management includes fire as a biomass control agent to reduce threats of biomass accumulation and watering plants during times of drought. We assume that grazing of stock will cease where MFL populations are protected under the MSA program.
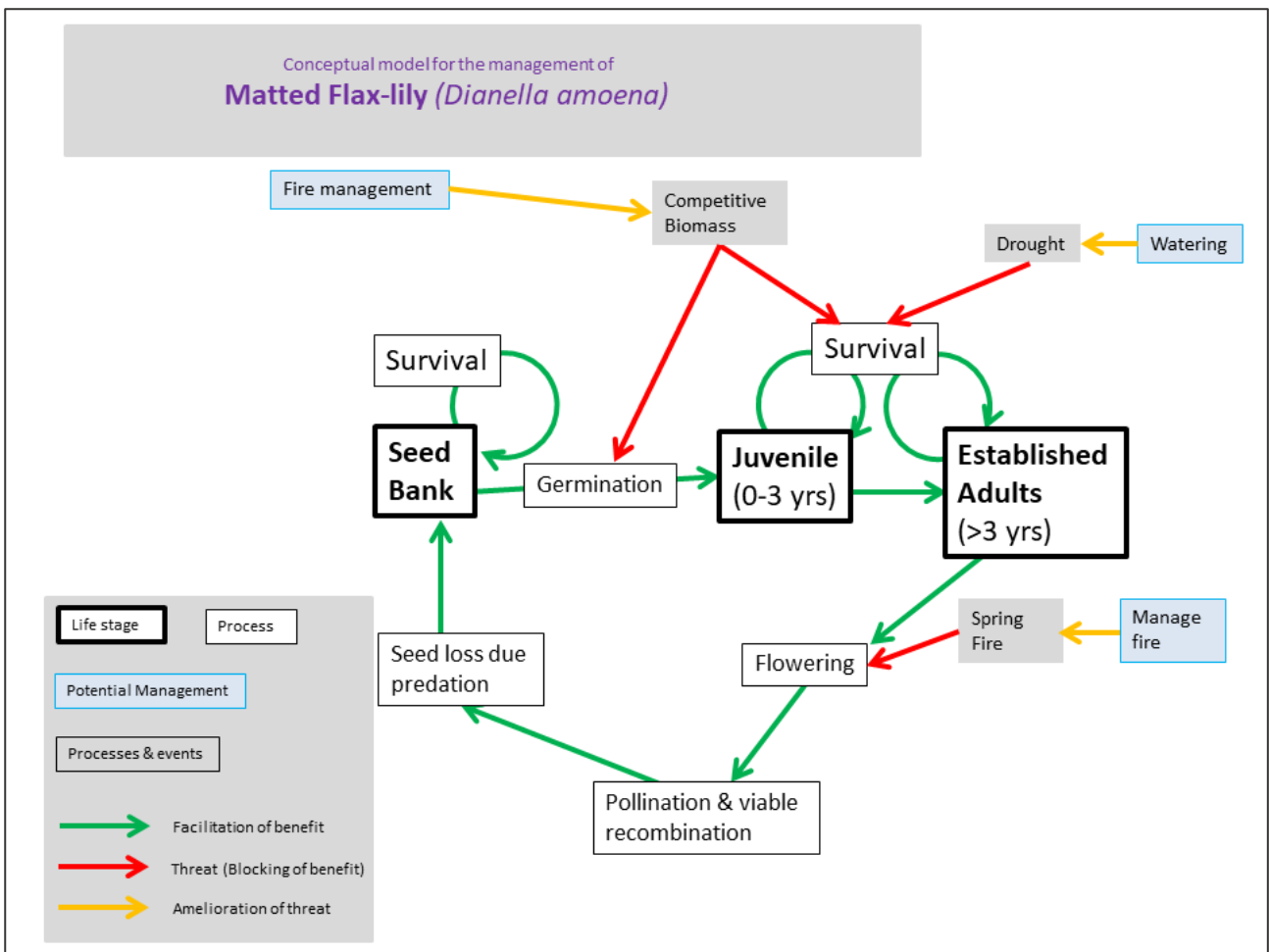
Figure 4.1: Conceptual model of the life cycle of the Matted Flax-lily including how threats and management influence different stages of the life cycle.

The timeline of important reproductive events and responses to fire seasons provides context for the way burning can ameliorate the threats of biomass accumulation (Figure 4.2). Burning has positive and negative effects on survival. In the short term there may be immediate mortality from fire but in the long term the survival rates are higher for those plants that survive fire or germinate after a fire due to reduced competition. Spring fire removes reproductive outputs for the year if it coincides with the flowering season. Fire after flowering season is optimal given plants are often dormant and below ground during summer and autumn (Figure 4.2).

Unofficial
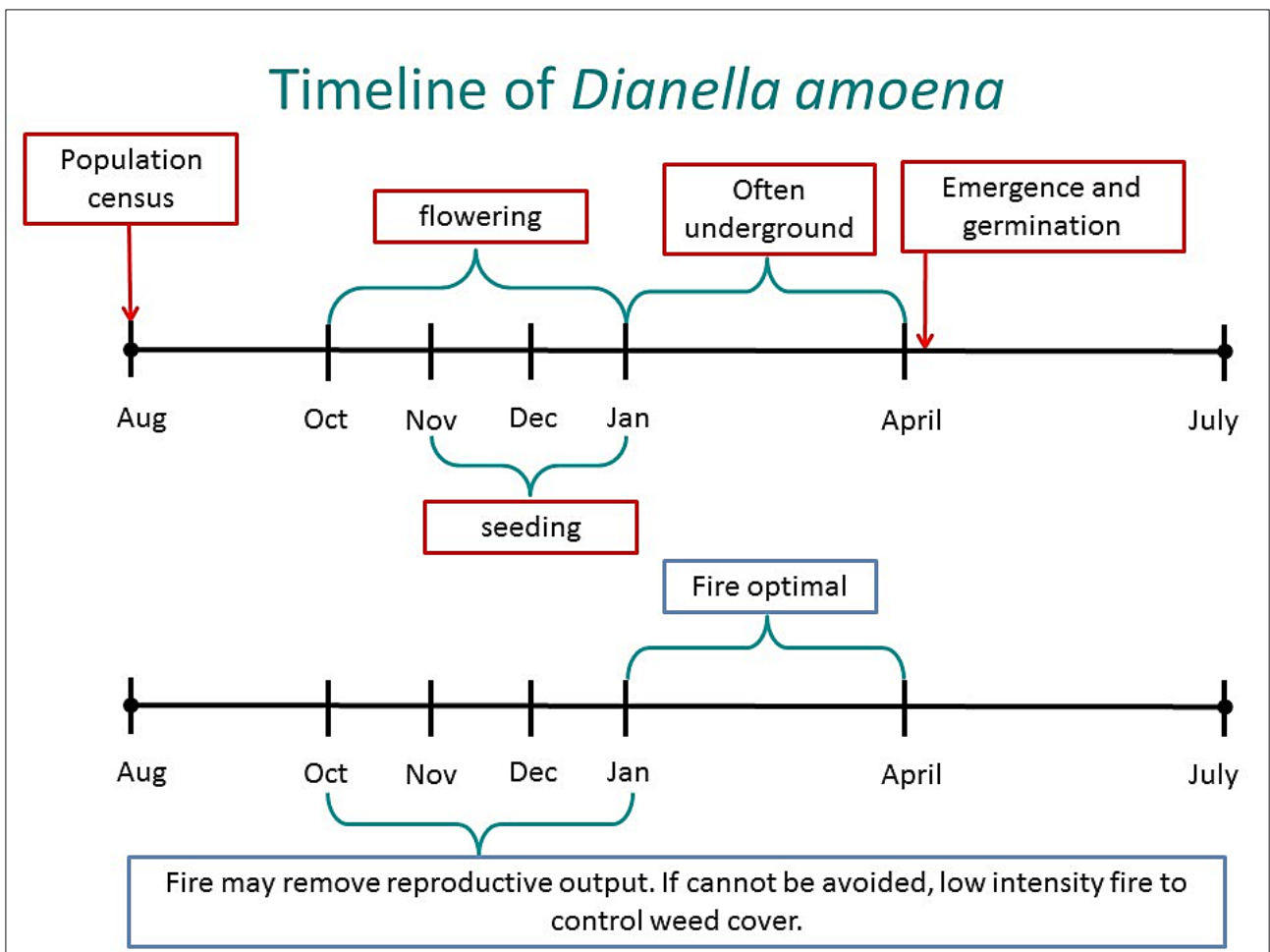
Figure 4.2: Conceptual model of the timing of ecological processes and effects of biomass control in different seasons

4.2.2 Vital rates

MFL was modelled using an age/stage class model, including a seedbank stage, three juvenile age classes and two reproductive adult stages. MFL plants to not reach reproductive maturity until they are older than three years. The three juvenile age classes ensure individuals do not progress to reproductive stages too quickly (i.e. Juvenile-adult). Two adult stages were modelled to reflect different survival rates of reproductive plants.

Survival rates

Survival rates for each of the age/stage classes were all derived from expert elicitation workshops. Survival rates assume low biomass conditions and represent the annual proportion of individuals surviving from one stage to another (Table 4.1). The experts estimated that recruitment in low biomass is 0.67, annual survival rate of recruits under the same conditions is 0.057 and survival rates of juvenile stages (1-3) is 0.105. Survival of adult stage 2 is 0.985.

Fecundity

The number of seeds an adult plant produces in a year is estimated to be 194.2 for each adult stage. Effective fecundity rate was then calculated by combining multiple processes that were elicited separately (i.e. age of first reproduction, proportion of seed loss from predation, proportion of viable seeds, seedbank longevity, fecundity increase with age etc) to this base rate. The mean values of these parameters were multiplied to give an estimate of the number of plants produced per preproducing individual that survives to their first census (August).

The growth rate used in the base model (no biomass control) was 1.0196. This growth assumes no droughts, low biomass conditions and no rainfall recruitment pulse.

Table 4.1: Life stage parameters for the Matted Flax-lily PVA model

Parameter	Value	Description
Adult 1 fecundity	194.2	Annual number of viable seeds per plant
Adult 2 fecundity	194.2	Annual number of viable seeds per plant
Seedbank survival	0.67	Annual proportion of seeds surviving each year
Seedbank to Juv 1	0.057	Annual proportion of seeds germinating each year and surviving to the census
Juv 1 to Juv 2	0.105	Annual proportion of 1st year juveniles surviving to their 2nd year
Juv 2 to Juv 3	0.105	Annual proportion of 2nd year juveniles surviving to their 3rd year
Juv 3 to Adult 1	0.105	Annual proportion of 3rd year juveniles surviving and transitioning to become stage 1 adults.
Adult 1 survival	0.885	Annual proportion of stage 1 adults surviving to the next year
Adult 1 to Adult 2	0.1	Annual proportion of stage 1 adults surviving and transitioning to become stage 2 adults
Adult 2 survival	0.985	Annual proportion of stage 2 adults surviving to the next year

4.2.3 Stochasticity

The model includes both environmental and demographic stochasticity. Default values for standard deviations on vital rates were assumed for all the plant models where standard deviations could not be estimated from the data. For fecundity estimates, standard deviations were derived based on a co-efficient of variation (CV) of 10% of the mean fecundity rate. For survival of recruits and juveniles, a CV of 10% was used. For adult survival, a CV of 0% was used because once plants reach adult stages, their survival is thought to be consistently stable. These assumptions were evaluated as part of the sensitivity analysis. Once empirical data on variability in vital rates become available, the new estimates can be incorporated into the model. The model assumes drought occurs every 20 years, with survival multiplier of 0.61 for adults and 0.48 for young adults in drought years.

Unlike many of the other grassland species in the MSA, MFL is not thought to have a recruitment pulse when rainfall is high during the autumn break. Thus all variation due to changes in rainfall are assumed to be captured in the standard deviations of the vital rates.

4.2.4 Biomass accumulation

Biomass accumulation can have adverse effects on Matted Flax-lily population due to increased competition for resources such as light and space. This phenomenon was modelled as a temporal trend that relates to the time since biomass control on x-axis as a function of annual survival rate of the species (Figure 4.3). Values of survival rates for juvenile and adult stages at the start and end of the five-year period since biomass control were elicited. Detrimental impact of biomass accumulation is only thought to begin after 15 years approximately, much longer than other grassland species. To model the time lag of impact as a function of biomass accumulation we used a step-like relationship (Figure 4.3). A non-linear version of this step-like relationship was investigated as part of the sensitivity analysis (Figure 4.4). Direct effects of biomass accumulation in the PVA were considered on two independent parameters; fecundity and survival rates.

Unofficial
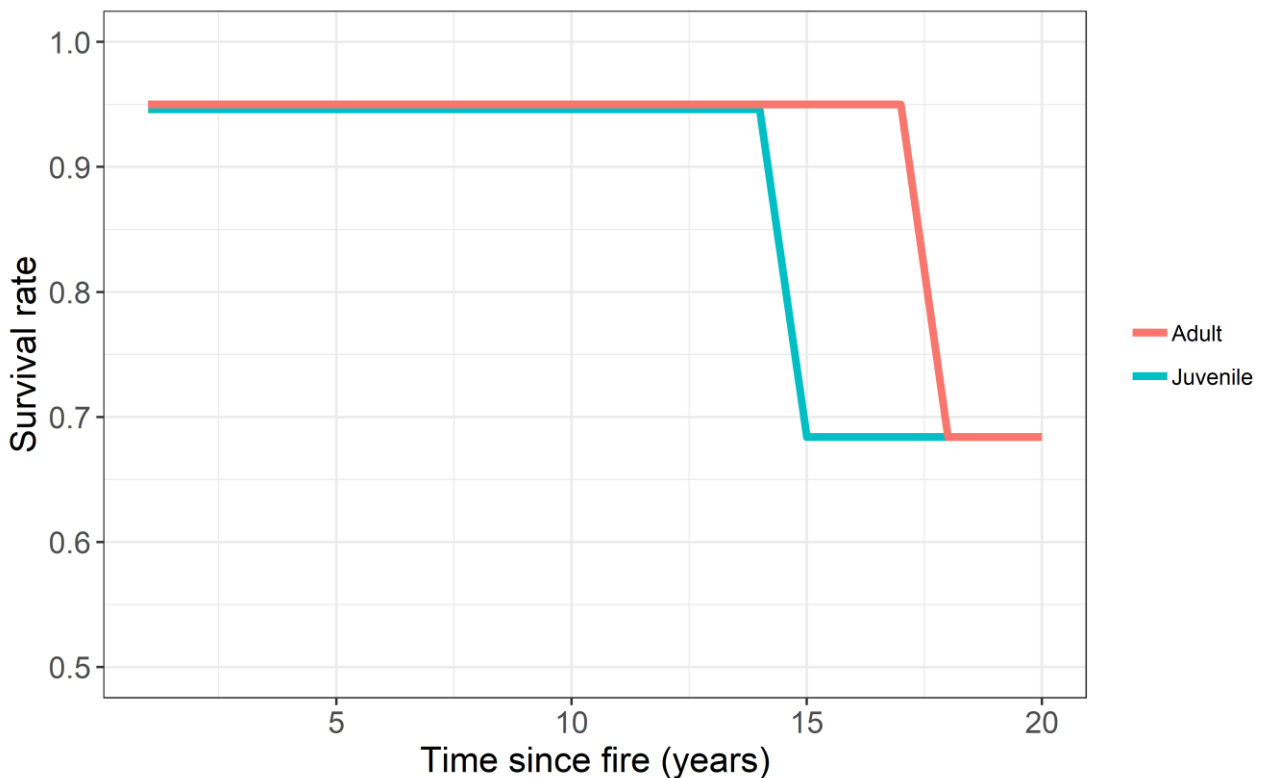
Figure 4.3: Relative change in survival rate of Matted Flax-lily in relation to time since biomass control (TSBC)

4.2.5 Additional model specification

Density dependence

A ceiling density dependence function is assumed across all stages. This means that modelled populations can grow exponentially until they reach a limit (ceiling) and then remain at that level. The carrying capacity was set to 200000, at a level that is high enough for ceiling capacity not to be reached by the model's timespan. The population growth was regulated through relationship between biomass accumulation (when increased, reduces growth rate) and controlled burning (when increased, increases growth rate).

Initial conditions

Initial population size was set at 10000. The stable age distributions with 10000 individuals are: seedbank = 9495; juvenile stage 1 = 403; juvenile stage 2 = 30; juvenile stage 3 = 22; adult stage 1 = 41; adult stage 2 = 9. The model was simulated 1000 times over 50 time-steps (1 time-step = 1 year). Model scenarios were compared using expected minimum abundance (EMA), the average of the minimum values for each trajectory of a stochastic population model (=1 simulation).

4.2.6 Management scenarios

Ten management scenarios were investigated (Table 4.2), including the null or "do nothing" scenario. Elicited values of immediate mortality and reduced fecundity due to various types of biomass control (combination of fire season, interval etc) were incorporated into each of the scenarios. Fire reduces survival of juveniles and adults by 33% and 3% respectively. Spring fire removes fecundity for that year. In the models this is averaged out over the number of fires in spring for a scenario. Short fire interval has 0.25 fires in spring, medium interval has either spring or autumn fires and long interval has 0.5 fires in spring. A fecundity multiplier is employed to reflect these assumptions into the model. For example, fire fecundity multiplier is 1 when proportion of spring fire is 0, whilst it is lowest at 0.67 (reducing fecundity by 33%) when proportion of spring fire is 1.

Watering

We assume that watering of individual plants during a drought ameliorates the drought effect on the species.

Table 4.2: Summary of management scenarios considered in each model

Scenario number	Description	Fire interval	Fire season	Conditions	Watering regime
D1	No fire management, no watering in drought conditions	No fire management	No fire	Drought	no water
D2	Short fire interval, Spring/Autumn, no watering in drought conditions	Short	Spring/Autumn	Drought	no water
D3	Medium fire interval Autumn, no watering in drought conditions	Medium	Autumn	Drought	no water
D4	Medium fire interval Spring, no watering in drought conditions	Medium	Spring	Drought	no water
D5	Long fire interval, Spring/Autumn, no watering in drought conditions	Long	Spring/Autumn	Drought	no water
D6	No fire management, watering in drought conditions	No fire management	No fire	Drought	watering
D7	Short fire interval, Spring/Autumn, watering in drought conditions	Short	Spring/Autumn	Drought	watering
D8	Medium fire interval Autumn, watering in drought conditions	Medium	Autumn	Drought	watering
D9	Medium fire interval Spring, watering in drought conditions	Medium	Spring	Drought	watering
D10	Long fire interval, Spring/Autumn, watering in drought conditions	Long	Spring/Autumn	Drought	watering
A1	No fire management, no watering in average conditions	No fire	No fire	Average weather	no water
A2	Short fire interval, Spring/Autumn, no watering in average conditions	Short	Spring/Autumn	Average weather	no water
A3	Medium fire interval Autumn, no watering in average conditions	Medium	Autumn	Average weather	no water
A4	Medium fire interval Spring, no watering in average conditions	Medium	Spring	Average weather	no water
A5	Long fire interval, Spring/Autumn, no watering in average conditions	Long	Spring/Autumn	Average weather	no water

4.2.7 Sensitivity analysis

Three types of sensitivity analysis were investigated: deterministic, stochastic and scenario sensitivity. The scenario sensitivity analysis evaluated two key assumptions in the model to determine whether the rank order of fire management scenarios change compared the base scenarios. In all base scenarios, the

temporal trend in survival rates due to biomass accumulation was assumed to be a step-like relationship (see Figure 4.3). A polynomial function was investigated (Figure 4.4) to test the effect of biomass accumulation on survival rates. Secondly, the most sensitive parameter from the stochastic sensitivity (i.e. survival rates) was investigated to see whether that sensitivity resulted in a change in the rank order of fire management actions.
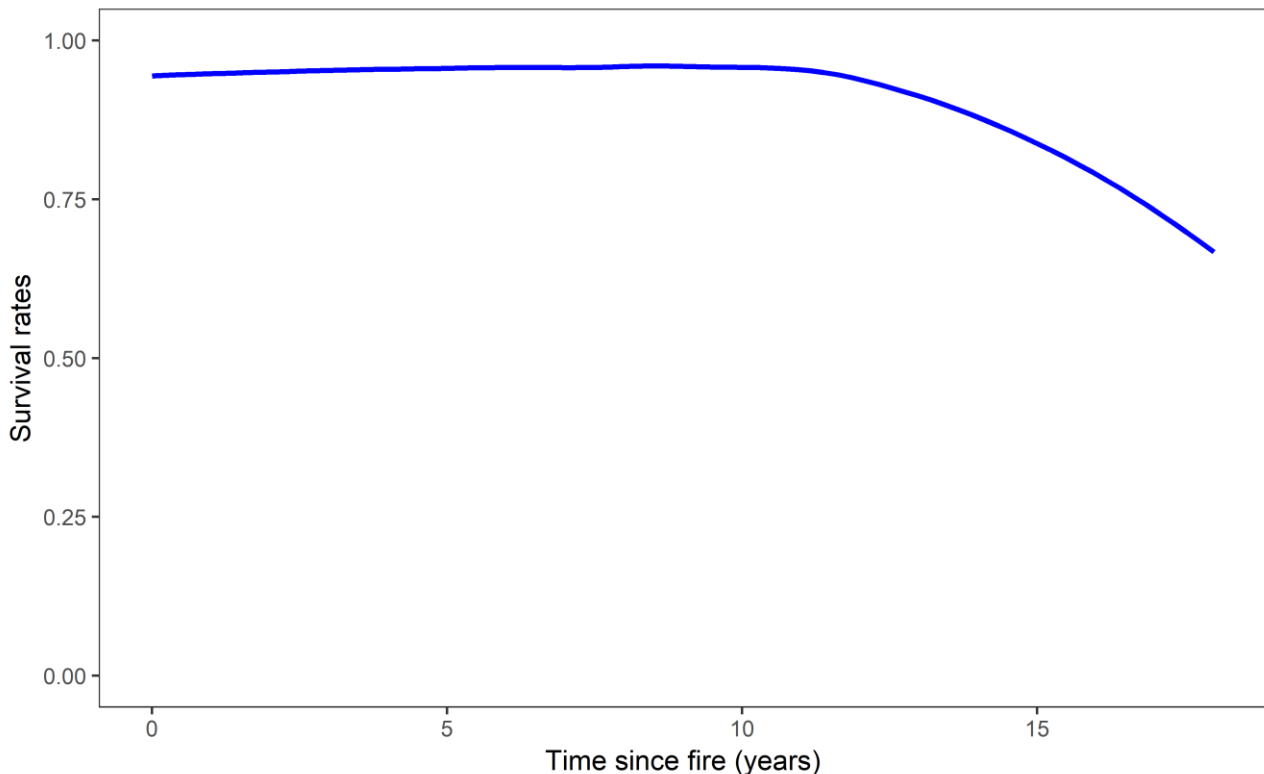
Figure 4.4: Example of non-linear relationship tested for trend in survival rates given TSBC used in scenario sensitivity analysis.

4.3   Results

4.3.1   Management actions

Results suggest competitive biomass needs to be managed in order for Matted Flax-lily to persist within the MSA management area into the future. In times of drought then watering plants will have benefits to persistence. For the fire scenarios investigated, medium interval autumn fires (i.e. burning every 3 years) were the most appropriate and resulted in the highest EMAs. Longer fire intervals (i.e. burning every 5 years) may also be beneficial in either fire season. The best four management scenarios all include watering, suggesting the importance of this management action to ameliorate the effects of drought.

No fire in the system can be detrimental for the species, the two management scenarios that didn't include fire (model #9 and #10 in Figure 4.5) return the lowest EMA amongst all the suites of management actions. When biomass accumulation is not controlled by fire, EMA is reduced by 95% compared to the best EMA achieved by medium interval autumn burn scenario. When watering is not part of the management suite, EMA is reduced by 60% even when the best fire management scenario (i.e. medium interval autumn burn) is employed. This suggests, without watering fire management is less effective.
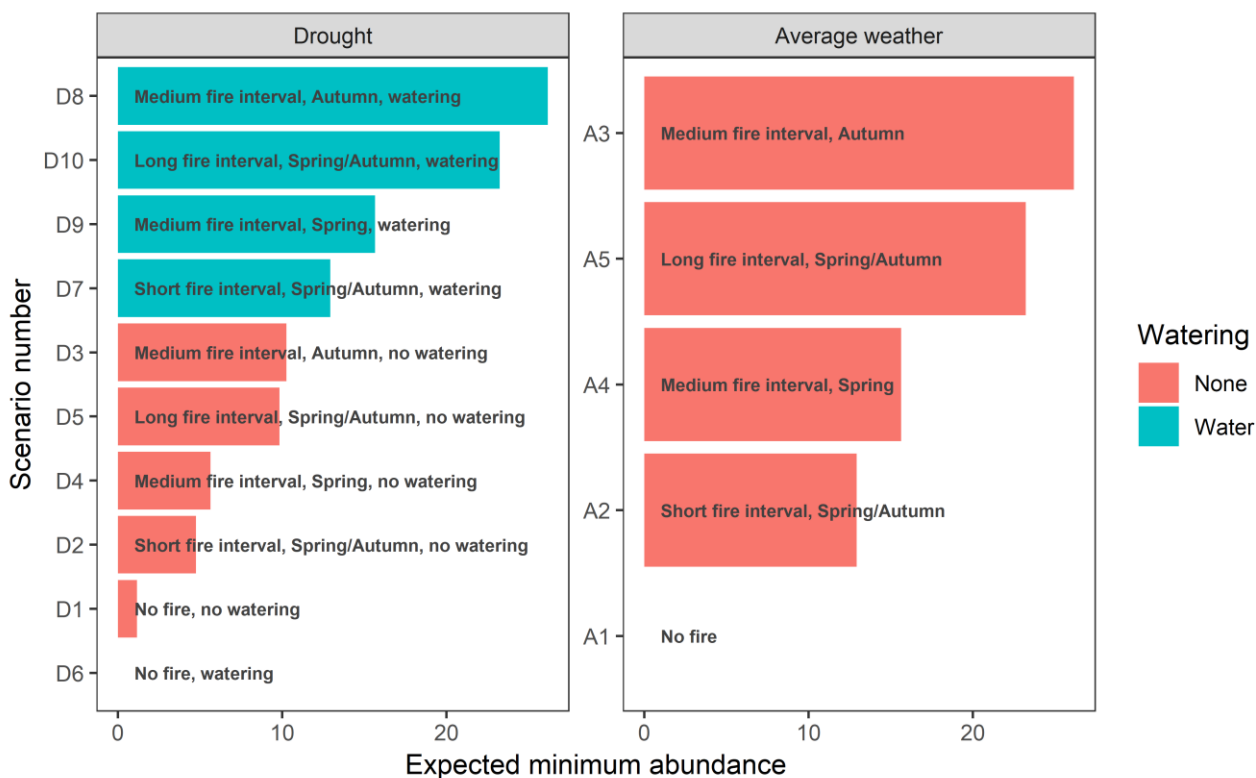
Figure 4.5: The result of PVA for Matted Flax-lily for drought and average weather. Ranking of management actions and subsequent EMA (excluding seedbank) for each scenario.

4.3.2   Sensitivity analysis

The deterministic sensitivity analysis revealed that adult survival is highly important for the growth rate. The proportional contribution of adult stages (2&1) survival to the growth rate is >60% and 20% respectively, with the other vitals rates providing small contributions (Figure 4.6). This was a similar result for the stochastic sensitivity analysis where positive and negative 10% changes in survival rates resulted in percent change in EMA of > 45% and >200% respectively (Figure 4.7). This suggests that understanding how adults are behaving under different threats and environmental conditions is vital for devising appropriate management actions for the species.

Scenario sensitivity analysis revealed that assumptions regarding the functional form (step-like) of the temporal trend in survival is sufficiently robust. The rank correlation between base models and the test scenario models (with polynomial function) is 0.564 (Table 4.3). The best and the worst management scenarios ranked the same for both functions, however, the rank order of other management scenarios varied slightly between the two functions. While reductions in survival had large changes in absolute values of EMA, the rank order of management actions had a correlation of 0.9, suggesting the assumed coefficient of variation in survival rate is reasonably reliable.
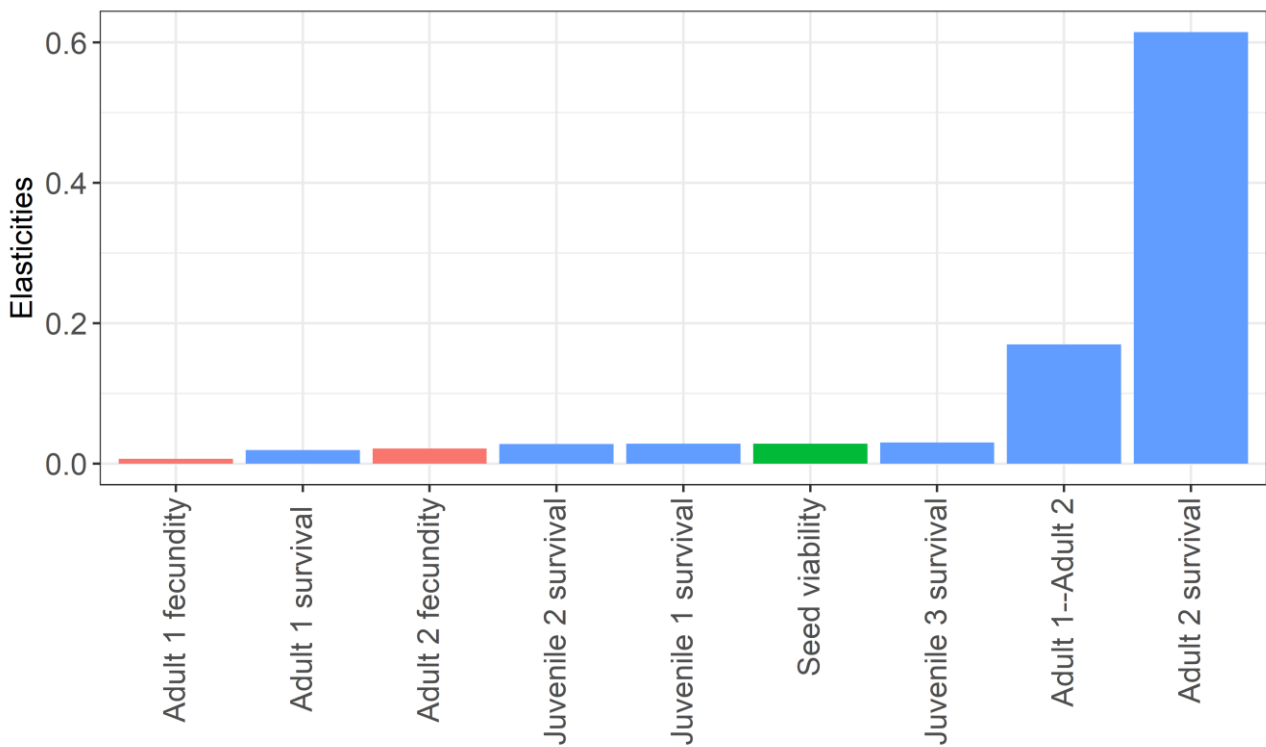
Figure 4.6: Result of deterministic sensitivity analysis. Elasticities of each life stage matrix parameter ranked in ascending order.
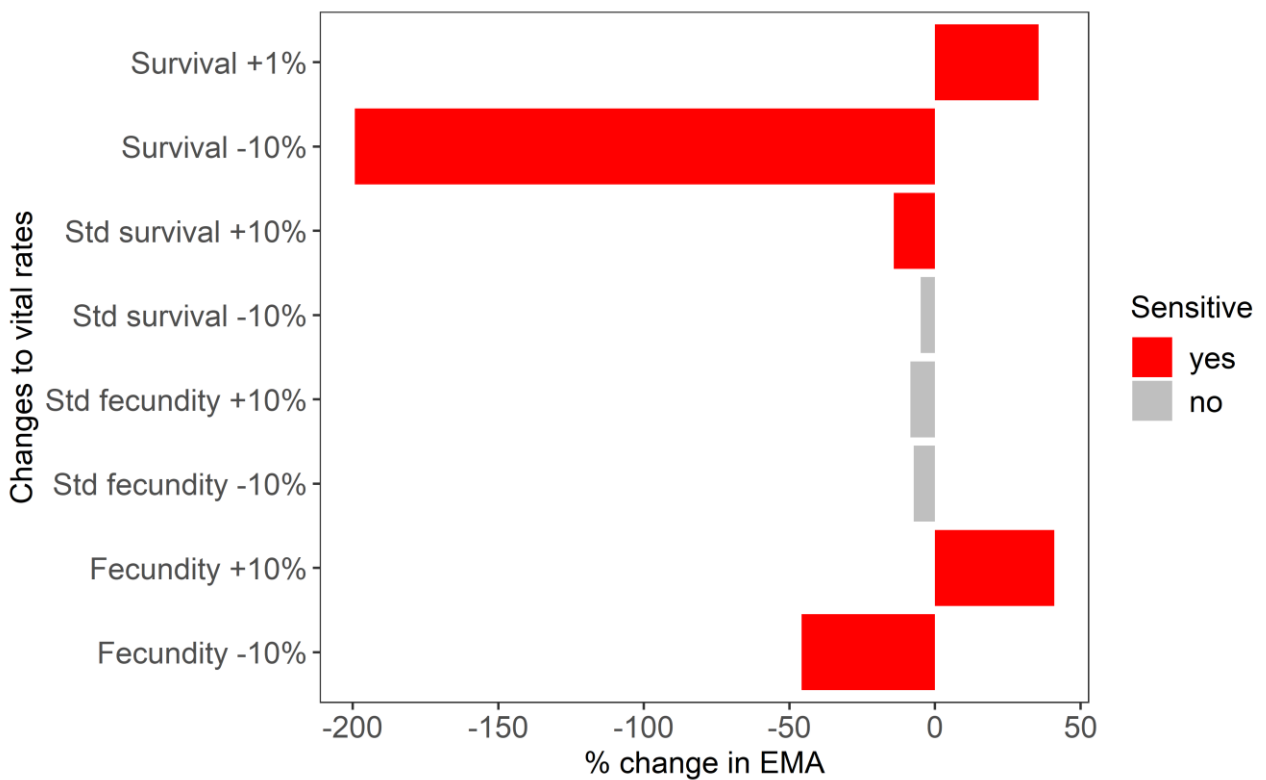
Figure 4.7: Result of stochastic sensitivity analysis. Changes in EMA greater than 10% (greater than input change) are coloured red.

Table 4.3: Scenario sensitivity analysis for MFL under drought conditions. EMAs and rank order of management actions. Rank correlations are with base models.

Management scenario	Base models	Polynomial function	10% reduction in survival
	EMA (Rank)	EMA (Rank)	EMA (Rank)
No fire management, no watering in drought conditions	1.194 (9)	1.032 (9)	0.004 (9)
Short fire interval, Spring/Autumn, no watering in drought conditions	4.687 (8)	2.068 (6)	0.015 (8)
Medium fire interval Autumn, no watering in drought conditions	10.549 (5)	3.11 (4)	0.046 (6)
Medium fire interval Spring, no watering in drought conditions	5.49 (7)	1.534 (7)	0.034 (7)
Long fire interval, Spring/Autumn, no watering in drought conditions	9.261 (6)	1.316 (8)	0.049 (5)
No fire management, watering in drought conditions	0.001 (10)	0.001 (10)	0 (10)
Short fire interval, Spring/Autumn, watering in drought conditions	12.497 (4)	5.957 (2)	0.055 (4)
Medium fire interval Autumn, watering in drought conditions	25.759 (1)	7.112 (1)	0.078 (2)
Medium fire interval Spring, watering in drought conditions	15.571 (3)	3.562 (3)	0.061 (3)
Long fire interval, Spring/Autumn, watering in drought conditions	22.812 (2)	2.993 (5)	0.079 (1)
Correlation coefficient		0.564	0.903

4.4 Discussion

We developed a population viability analysis model for Matted Flax-lily using multiple expert judgements to explore the effectiveness of several management options on the persistence of the species. This will inform the development of a management plan. Results suggest medium to long fire intervals (every 2-7 years) in Autumn and watering plants during times of drought is the best management action of those considered here. Without combining both actions, the effectiveness of any one action is substantially reduced. Before using these results to develop a management plan for the species, consultation with all experts involved in the elicitation will be required to ensure interpretation of model results is correct and recommendations for management are sound. Furthermore, the practicalities and feasibility of watering individual plants during times of drought will need to be assessed.

The sensitivity analysis revealed that adult survival is an important parameter influencing the model. While uncertainty in survival did not alter ranks of fire management scenarios, if absolute values of abundance are preferred, then better estimates of survival are necessary. Monitoring of MSA populations (DELWP 2015) is collecting longitudinal data on MFL and therefore we will be able to calculate survival rates. The large influence of adult survival on model behaviour suggests investigating an alternative model structure may be

warranted. For instance, including multiple adult stages may provide some extra nuance within the adult stage that may be important for the population dynamics and subsequent management. There is a trade off as this added complexity will require more information to parameterise.

The scenario sensitivity analysis investigated two model assumptions for trends in vital rates due to biomass accumulation. This analysis revealed the model is generally robust to our assumptions, at least for the scenarios tested. Changing the shape of the biomass/survival relationship change the ranks of some fire scenarios but did not alter the top and bottom ranks. It is likely the model will be sensitive to other assumptions that were not tested. In the future it is worthwhile exploring a more detailed sensitivity analysis to fully explore the behaviour of the model with respect to other important assumptions that were not considered here.

The model for this species is based purely on expert judgement as no empirical data is currently available for the species. To improve our understanding of the species dynamics how it responds to alternative management options, basic life history data on survival and reproduction under different environmental conditions is needed. Consultation with species experts and researchers is required to develop a research plan for MFL on how to best capture the data needed to inform the model. Furthermore, we could update our monitoring plan in order to collect the required demographic data.

One of the largest risks for this species is the small population size, as a single unforeseen catastrophic event could cause it to go locally extinct. Small populations are also at risk of extinction due to Allee effects (Courchamp et. al. 2008). This aspect of the species potential fate was not considered in the model development or scenarios. While this study has provided some insights around management of fire and watering, for the ongoing persistence of the species it will be important to create more populations of MFL and or to increase the size of existing populations to spread the risk. Creating or enhancing populations through planting of established individuals or through direct seeding should be considered as part of the management plan for the species. Translocations have been successfully conducted in the past (Carr and Rodda 2011). The PVA model developed here could be extended and adapted to explore scenarios of population creation and evaluate options for how it can be achieved.

5 Small Golden Moths Orchid (*Diuris basaltica*)

5.1 Background

The Small Golden Moths Orchid (*Diuris basaltica)* is a deciduous, perennial orchid endemic to Victoria. It grows in herb-rich native grasslands dominated by Kangaroo Grass (*Themeda triandra)* on heavy basalt soils. It has linear leaves and produces one or two small (~20mm) golden flowers that emerge annually in September and October (Jones 2006). The species survives the dry summer as a dormant tuber that is replaced annually. Reproduction is from seed and vegetative reproduction through tuber multiplication (Backhouse and Lester 2010).

The Small Golden Moth Orchid (SGMO) is restricted to a small number of populations in the west of Melbourne stretching from Sydenham in the north to Lara in the south, a distance of about 50 km, in the Victorian Volcanic Plain IBRA bioregion (Backhouse and Lester 2010). Habitat loss from converting grasslands to agriculture and urban and industrial development in Melbourne's west, has been the major cause of decline of SGMO across its distribution. There is a high risk of local extinction of remaining populations due to the small population sizes at the remaining sites (Backhouse and Lester 2010).

A population of approximately 400 individuals of SGMO is found within Conservation Area 3 within the MSA management area. This population is the focus of the development of population viability analysis model for the species. Very little is known of the biology and ecology of SGMO with little to no empirical evidence available. Thus, the PVA model was developed based solely on the judgement of experts to explore the effectiveness of alternative management options on the persistence of the species within the MSA management area. The focus of SGMO as part of the MSA program and the insights from the model process regarding important knowledge gaps, aim to help guide data collection, provide improvements to the PVA model, refine our understanding of the population dynamics of the species and subsequently improve management of the species over time.

5.2 Methods

5.2.1 Conceptual models

Two conceptual models for the SGMO were developed in collaboration with species experts. Firstly, a conceptual model of the life cycle of the species and how threats and management influence important life stages (Figure 5.1). The species has been described in two main stages, seedlings and adults. Plants remain as seedlings for up to three years before moving into the adult stage. Plants can become reproductive at three years of age but at much lower rates than adult plants that are assumed to be >3 years of age. Seeds are thought to survive for one year, thus we assume there is no viable seed bank for the species.

The main threats to the SGMO within the MSA management area are competitive biomass, herbivory from rabbits, prolonged drought and inappropriate burning regimes. The accumulation of biomass over time is thought to reduce the survival of all plants but at different rates depending on the age of the plant. While fire is a useful management action for reducing the threat of biomass accumulation, it can increase mortality of young plants and if done while flowering and seeding, can remove the reproductive output for the year. Rabbits reduce both survival and reproductive output as rabbits' graze on both seedlings and flowering adults. Prolonged drought increases mortality of plants especially seedlings. It is thought there is a complex relationship between herbivory and biomass accumulation on reproductive output (M. Duncan, *pers. comms.*). Herbivory of flowers is thought to be higher when biomass is low, when flowers are easily seen and there is less biomass to graze. Once biomass increases, less flowers are browsed upon as they are less conspicuous to rabbits.

The main management options for SGMO include reducing competitive biomass through burning, protecting plants from herbivory by constructing rabbit proof fences, and watering plants during times of drought. Burning has positive and negative effects on survival; there is an increase in mortality immediately following the fire but for those plants that survive fire, or germinate after a fire, have improved survival rates due to

Unofficial

reduced competition (Figure 5.1). We assume no grazing will occur in populations of SGMO under MSA management.
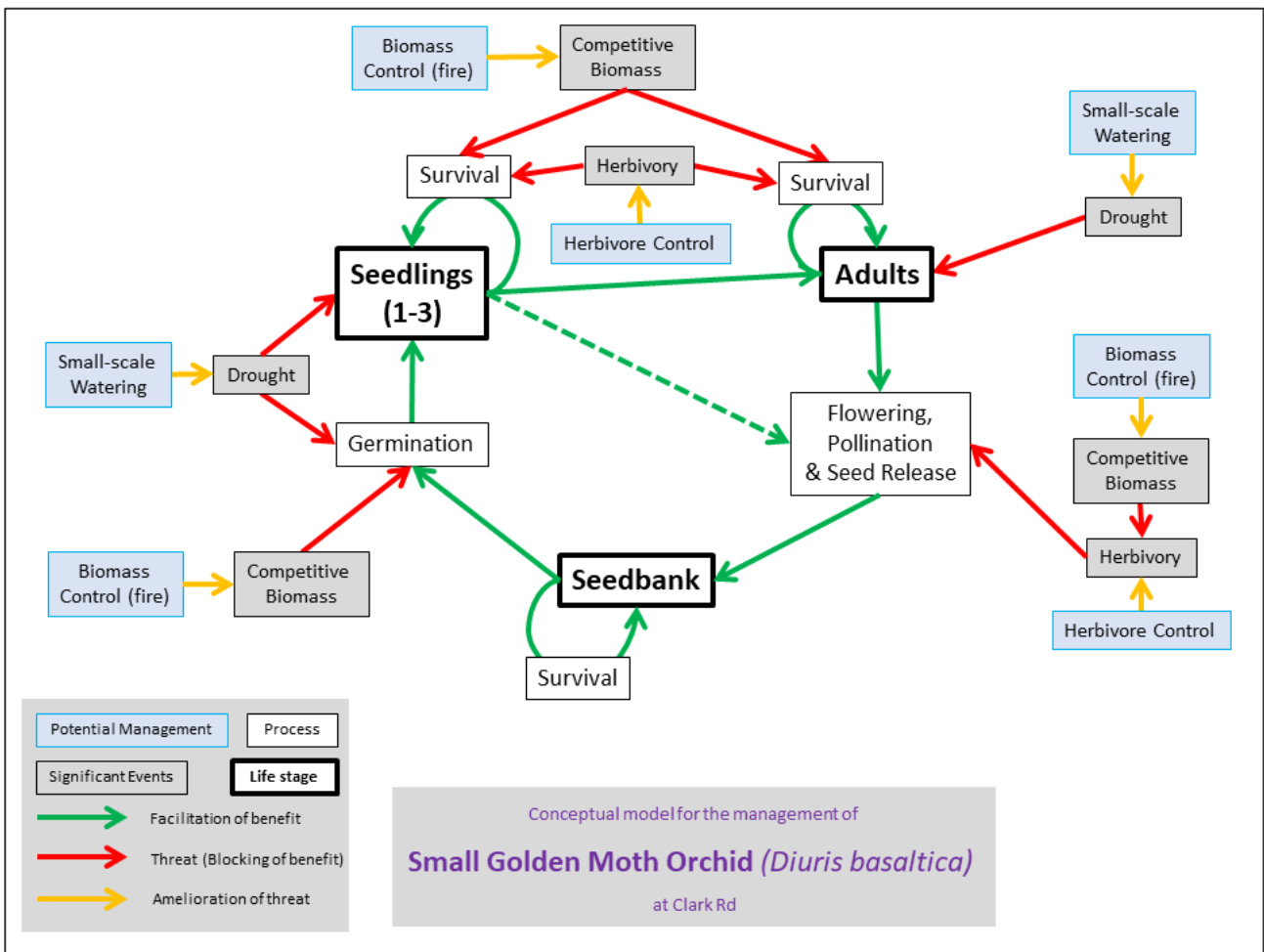
Figure 5.1: Conceptual model of the life cycle of the Small golden moth orchid including how threats and management influence different stages of the life cycle.

A second conceptual model illustrates the timing of different processes over a year and the impact of the timing of fire on survival and reproduction (Figure 5.2). Adults emerge between March and May, and tuber replacement is thought to occur between April and August. Flowering and seed production generally occurs between August and October. Plants enter the annual dormancy phase in late spring, seedlings first, followed by adults. The timing of germination is unknown. If fire occurs over the summer months when plants are in their dormant phase, there is little to no increase in mortality. Fire between February-April can induce small increases in mortality. Fires in winter and spring can increase mortality especially younger plants. Reproductive output is reduced if fires occur during flowering and seed production predominantly during August-October (Figure 5.2).
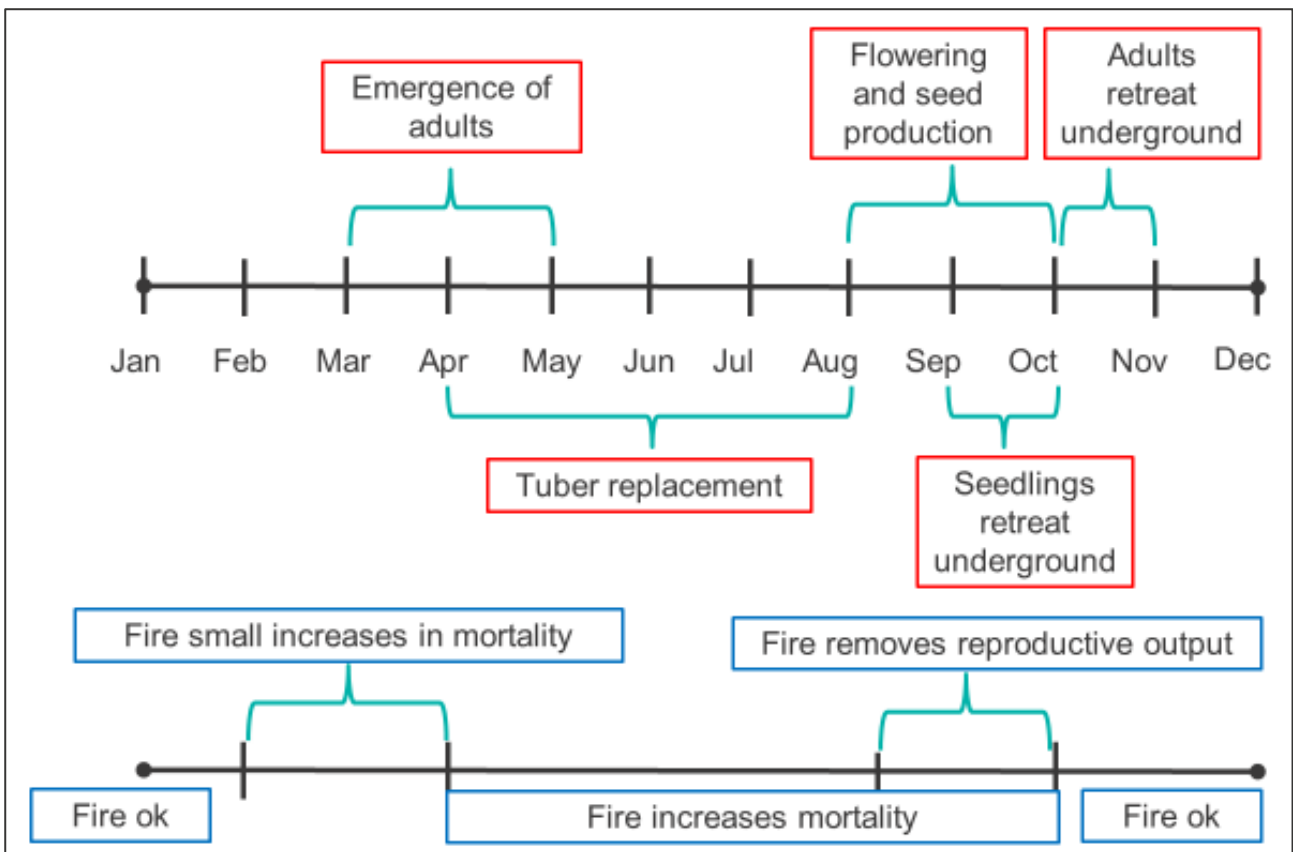
Figure 5.2: Conceptual model of the timing of ecological processes and effects of biomass control in different seasons.

5.2.2   Vital rates

The population of SGMO was modelled using an age and stage class model where seedlings were modelled as three separate age classes (i.e. year 1 to year 3) and adults were modelled as one stage class, where individuals can remain as an adult for multiple years. A structured expert elicitation workshop was run to elicit judgements on all model parameters. Survival and fecundity of the orchid was estimated under different scenarios regarding the presence or absence of drought and rabbits (Table 5.1). The calculation of fecundity includes the combination of multiple processes that were elicited separately, including, percentage of plants that emerge, proportion of plants that flower, number of flowers per flowering plant, seeds per capsule, seed viability, germination rate and survival to first census. The mean values of these parameters were multiplied to give an estimate of the number of plants produced per reproducing individual that survive to the first census (Table 5.1).

Models were calibrated to ensure the resulting growth rate was within the bounds of the maximum growth rates elicited from the experts and an average life span of approximately 30 years. A species expert (M. Duncan, ARI) was consulted during model development to provide advice on model behaviour.

Unofficial

Table 5.1: Life stage parameters for Small Golden Moths Orchid under different rabbit and drought combinations.

Parameter	Value				Description
	No rabbits, no drought	Rabbits, no drought	No rabbits, drought	Rabbits & drought	
Seedling 3 fecundity	0.3	0.058	0.00155	0.0003	The number of new recruits per 3-year old seedling each year
Adult fecundity	0.77	0.148	0.00416	0.0008	The number of new recruits per adult each year
Seedling 1 to Seedling 2	0.27	0.24	0.05	0.044	Annual proportion of 1st year seedlings surviving to their 2nd year
Seedling 2 to Seedling 3	0.46	0.39	0.4	0.34	Annual proportion of 2nd year seedlings surviving to their 3rd year
Seedling 3 to Adult	0.722	0.68	0.69	0.64	Annual proportion of 3rd year seedlings surviving to adulthood
Adult survival	0.965	0.96	0.95	0.947	Annual proportion of adults surviving to the next year
Growth rate (λ)	1.0304	0.97	0.95	0.947	

5.2.3 Stochasticity

The level of natural variation in survival and fecundity is unknown and was not elicited. Instead we have used default values that were assumed for all the plant models where standard deviations could not be estimated from data. For fecundity estimates, standard deviations were derived based on a coefficient of variation (CV) of the mean vital rates of 20%. For survival rates, a CV of 10% was used. This assumption was evaluated as part of the sensitivity analysis. As data on variability in vital rates comes to hand, improved estimates can be incorporated into the model.

5.2.4 Biomass accumulation

The accumulation of biomass around SGMO plants is thought to reduce survival rates due to competition for light and space. This phenomenon was modelled as a temporal trend that relates the time since biomass control (TSBC) and the annual survival rate of the species. End points for survival rates under low and high biomass conditions were elicited from experts and a linear relationship between these end points was assumed (Figure 5.3). Non-linear relationships were also investigated as part of the sensitivity analysis. The time since biomass control where biomass was considered high, was inferred from available data collected at Truganina Cemetery, Mount Cottrell and Kalkallo Common (DELWP 2015). Similar functions were derived for the drought scenarios with and without rabbits.
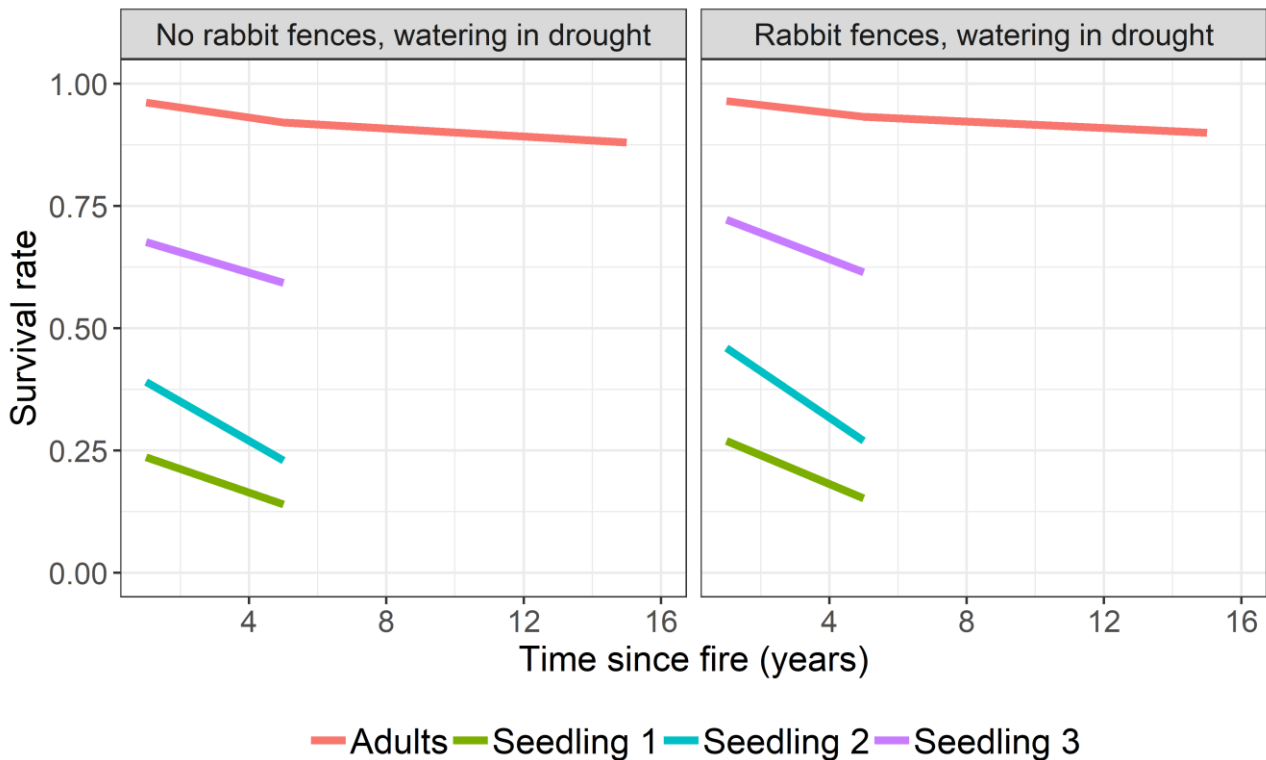
Figure 5.3: Relationship between survival of different life stages and time since biomass control (TSBC) in years with and without management of rabbits.

There was disagreement among the experts as to whether germination rates were affected by biomass accumulation. For this version of the model, constant rates have been assumed under high and low biomass.

When rabbits are in the system, fecundity is reduced through herbivory of flowers. As biomass accumulates, rabbits find it more difficult to find flowers, or have other resources available so the loss of flowers reduces. This effect is thought to be pronounced in drought conditions. This was modelled as a positive temporal trend in fecundity as a function of TSBC. Evaluation of this assumption was investigated as part of the scenario sensitivity. Estimates for percentage reduction in flowers due to herbivory from rabbits were elicited from one of the species experts and incorporated into the fecundity estimates (Table 5.2). A linear relationship between low and high biomass was assumed.

Table 5.2: Percent reduction in flowers due to herbivory from rabbits under low and high biomass and drought and no drought conditions

	Low Biomass	High Biomass
Average conditions	75%	40%
Drought conditions	90%	75%

5.2.5   Additional model specification

Density dependence was incorporated as a ceiling model. In this type of model populations can grow exponentially according to the stage matrix until they reach a ceiling, and then remain at that level. Carrying capacities were set high so that the ceilings could not be reached within the timespan of the model. Instead populations were regulated through the relationship between vital rates and biomass accumulation (reduced capacity to grow), and controlled burning (increased capacity to grow).

The initial abundance was set at 400 individuals to reflect the current known population and assumed to be at a stable age distribution. The model was run over 50 years for 1000 iterations. Model scenarios were compared using expected minimum abundance (EMA).

Unofficial

5.2.6 Management scenarios

The five standard fire management scenarios (Table 1.1) were applied to each of the four model combinations that included the presence and absence of drought and rabbits, resulting in 20 alternative management scenarios (Table 5.3). Immediate mortality and reduced reproduction due to fire and fire season were elicited from experts and applied to each of the scenarios. If fires occur in autumn before the autumn break, immediate mortality of plants in any of the seedling stages is 38% in the year the fire occurs. If fire occurs in spring, immediate mortality of seedlings increases to 58% in the year the fire occurs. In addition, 91% of reproductive output for that year is lost in a spring fire. It is assumed there is no immediate mortality of adult plants due to fire in any season.

The "*no rabbit*" scenarios assume rabbits are managed and removed from the system through the construction of exclusion fences. Scenarios around watering during times of drought were not explicitly modelled. We assumed the effect of watering during a drought increases the survival rates to the no drought rates. Thus, the effect of watering during drought conditions is the difference between the drought and no drought scenarios. Drought scenarios assumed drought conditions across the entire time frame, which may be unrealistic. However, modelling constraints prevented us from including stochastic droughts in the model as multi-year episodic events during the 50-year timeframe. Despite this limitation, insights of the effect of watering under drought conditions on the persistence of SGMO can still be inferred from the many scenarios investigated.

Table 5.3: Summary of management scenarios considered in each model

Scenario	Description	Fire interval	Fire season	Rabbit exclusion fence	Conditions
1	No fire management, rabbit exclusion fence, drought conditions	No fire management	NA	Present	Drought
2	Medium fire interval, fire in Autumn, rabbit exclusion fence, drought conditions	Medium	Autumn	Present	Drought
3	Long fire interval, fire in Autumn/Spring, rabbit exclusion fence, drought conditions	Long	Autumn/Spring	Present	Drought
4	Short fire interval, fire in Autumn/Spring, rabbit exclusion fence, drought conditions	Short	Autumn/Spring	Present	Drought
5	Medium fire interval, fire in Spring, rabbit exclusion fence, drought conditions	Medium	Spring	Present	Drought
6	No fire management, rabbit exclusion fence, average conditions	No fire management	NA	Present	Average condition
7	Medium fire interval, fire in Autumn, rabbit exclusion fence, average conditions	Medium	Autumn	Present	Average condition
8	Long fire interval, fire in Autumn/Spring, rabbit exclusion fence, average	Long	Autumn/Spring	Present	Average condition

	conditions				
9	Short fire interval, fire in Autumn/Spring, rabbit exclusion fence, average conditions	Short	Autumn/Spring	Present	Average condition
10	Medium fire interval, fire in Spring, rabbit exclusion fence, average conditions	Medium	Spring	Present	Average condition
11	No fire management, no rabbit exclusion fence, drought conditions	No fire management	NA	Absent	Drought
12	Medium fire interval, fire in Autumn, no rabbit exclusion fence, drought conditions	Medium	Autumn	Absent	Drought
13	Long fire interval, fire in Autumn/Spring, no rabbit exclusion fence, drought conditions	Long	Autumn/Spring	Absent	Drought
14	Short fire interval, fire in Autumn/Spring, no rabbit exclusion fence, drought conditions	Short	Autumn/Spring	Absent	Drought
15	Medium fire interval, fire in Spring, no rabbit exclusion fence, drought conditions	Medium	Spring	Absent	Drought
16	No fire management, no rabbit exclusion fence, average conditions	No fire management	NA	Absent	Average condition
17	Medium fire interval, fire in Autumn, no rabbit exclusion fence, average conditions	Medium	Autumn	Absent	Average condition
18	Long fire interval, fire in Autumn/Spring, no rabbit exclusion fence, average conditions	Long	Autumn/Spring	Absent	Average condition
19	Short fire interval, fire in Autumn/Spring, no rabbit exclusion fence, average conditions	Short	Autumn/Spring	Absent	Average condition
20	Medium fire interval, fire in Spring, no rabbit exclusion fence, average conditions	Medium	Spring	Absent	Average condition

5.2.7 Sensitivity analysis

Three types of sensitivity analysis were investigated: deterministic, stochastic and scenario sensitivity. The deterministic sensitivity analysis focused on the elasticities of the stage matrix, i.e. the proportional contribution of each of the parameter estimates toward the growth rate implied by the stage matrix. The stochastic sensitivity analysis involved adjusting each of the vital rates and standard deviations of a stage

matrix separately by ±10% to investigate if the stochastic model was sensitive to any these parameters. If the EMA results changed by more than ±10% this indicates the model is sensitive to the parameter.

The scenario sensitivity analysis evaluated three key assumptions in the model to determine whether the rank order of fire management scenarios change compared the base scenarios. The temporal trend in survival rates due to biomass accumulation was assumed to be linear. Two non-linear functions were investigated, a convex shaped curve and a concave shaped curve (Figure 5.4). The convex and concave functional forms were applied to all survival rates in model. The relationship between herbivory of flowers and biomass accumulation was also investigated. Base models were compared to models where fecundity remained constant through time. Lastly, the most sensitive parameter from the stochastic sensitivity was investigated to determine the influence on the rank order of fire management actions. All scenario sensitivity analysis was performed using the rabbit and no drought model.
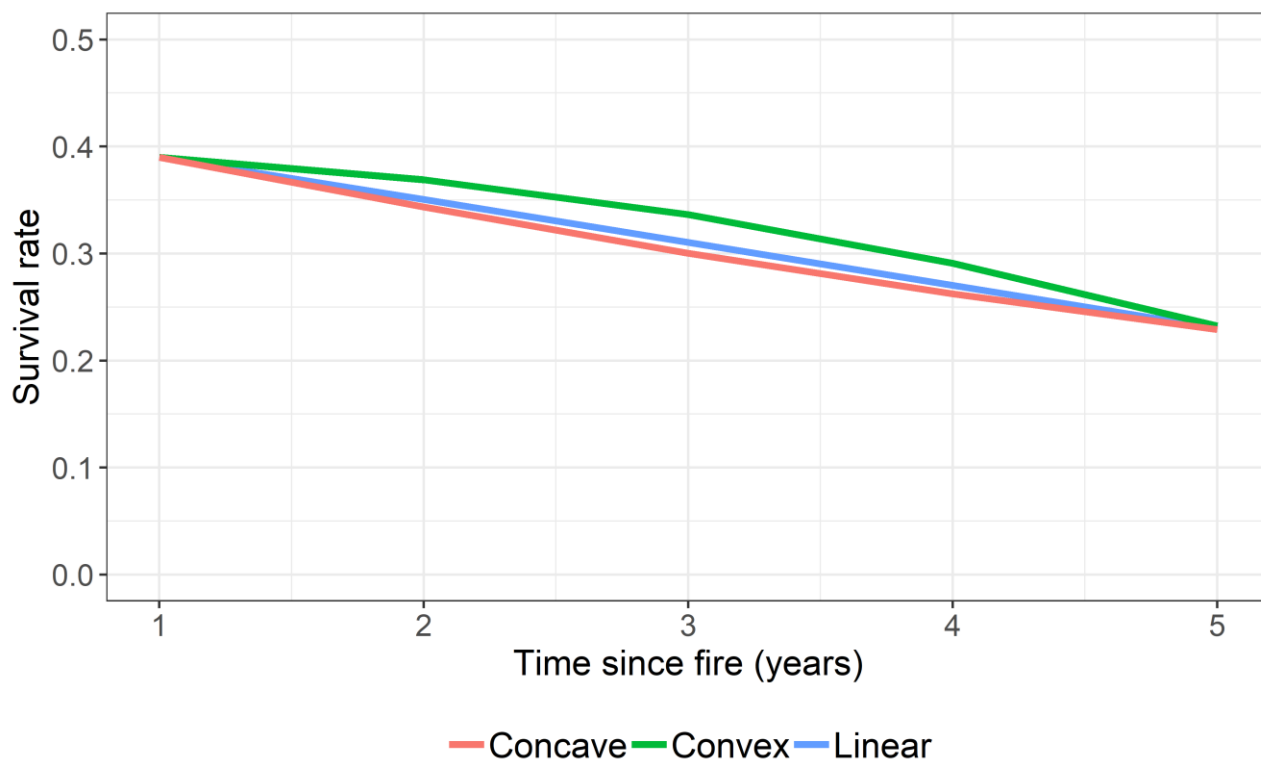
Figure 5.4: Example of different functional forms for trend in survival given TSBC used in the scenario sensitivity analysis

5.3 Results

5.3.1 Management actions

Results suggest all threats need to be managed simultaneously for continued persistence of the SGMO within the MSA management area (Figure 5.5). In the presence of multiple threats, no one action is sufficient to manage the species. For the fire scenarios investigated, medium interval autumn fires were the most appropriate. Medium fire intervals refer to burning every 3 years. Shorter fire intervals (i.e. burning every 1-3 years) may also be beneficial but given they may occur in spring, resulted in a larger cost to the species due to higher immediate mortality and loss of reproductive output. As fire intervals become longer, the EMA decreases. The absence of fire in the system can be catastrophic for the species, particularly if other threats are not managed. These results were consistent across the four model combinations that included the presence and absence of drought and herbivory from rabbits.

Uncontrolled rabbits are predicted to have a large impact on the persistence of the orchid. When rabbits are not controlled, EMAs reduced by 80% even when fire was managed appropriately, suggesting without rabbit control, fire management is less effective. Similarly, prolonged droughts resulted in very low EMAs across all fire scenarios. Interpretation of drought scenarios needs to account for droughts being modelled over the entire 50-year time frame. However, results suggest that watering plants during drought conditions should be

part of the suite of management actions to protect the species and without it, other management actions may be less effective (Figure 5.5).
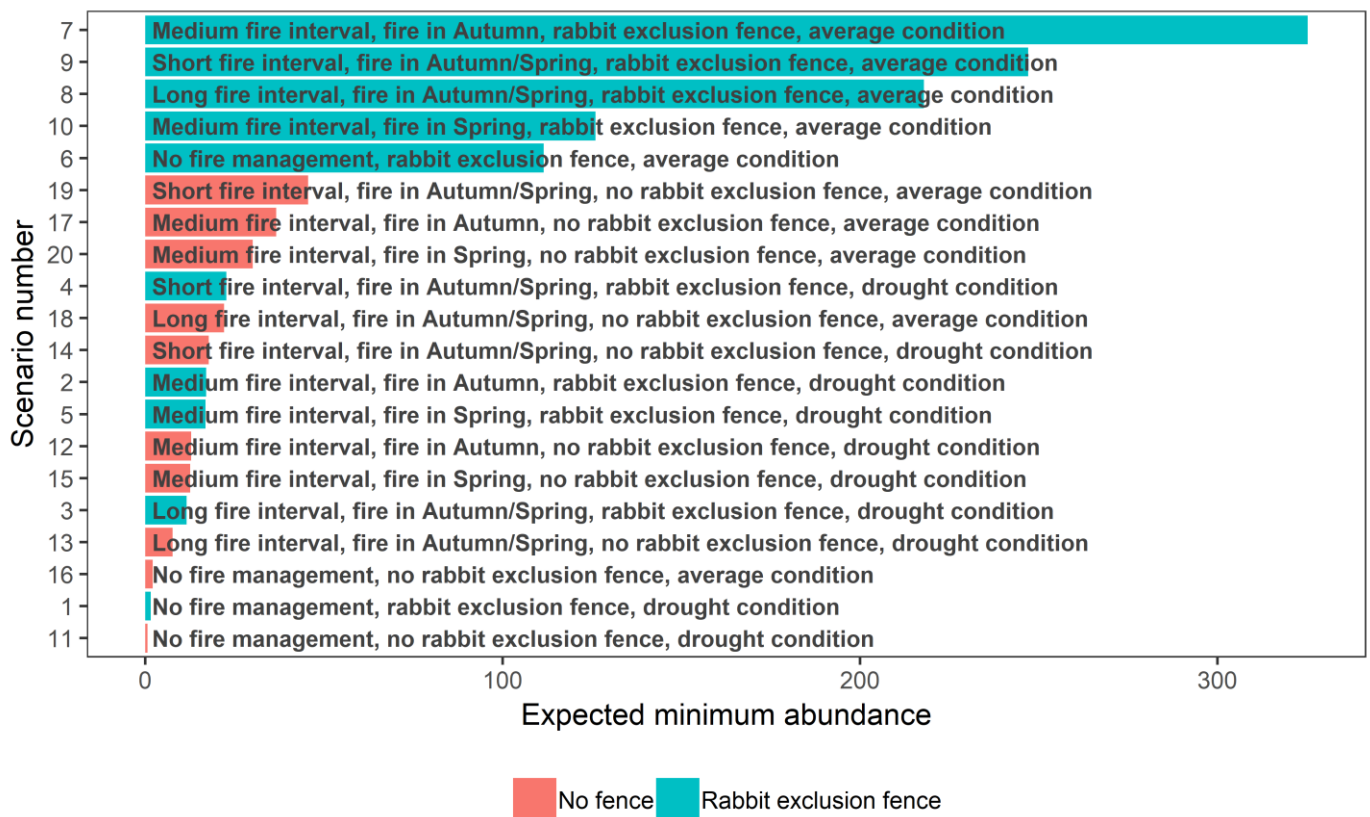
Figure 5.5: The result of PVA for Small Golden Moth Orchid– ranking of management actions and subsequent EMA for each scenario.

5.3.2   Sensitivity analysis

The deterministic sensitivity analysis revealed that adult survival has overwhelming influence on the growth rate. The proportional contribution of adult survival to the growth rate was >90% with the other vitals rates providing small contributions (Figure 5.6). This was a similar result for the stochastic sensitivity analysis where ±10% changes in survival rates, resulted in percent changes in EMA of >100% and >200% respectively (Figure 5.7). This suggests that understanding how adults are behaving under different threats and environmental conditions is vital for devising appropriate management actions for the species.

Scenario sensitivity analysis revealed preferred fire management for the species is robust to assumptions regarding the trend in survival, at least based on the alternative tested. The rank order of fire management scenarios was identical irrespective of form used (Table 5.4). For assumptions regarding increased herbivory of flowers in low biomass conditions, models that used a constant rate performed similarly to models with a temporal trend. Ranks slightly changed but had a correlation 0.8. While reductions in survival had large changes in absolute values of EMA, the rank order of management actions had a correlation of 0.87 (Table 5.4).

Unofficial
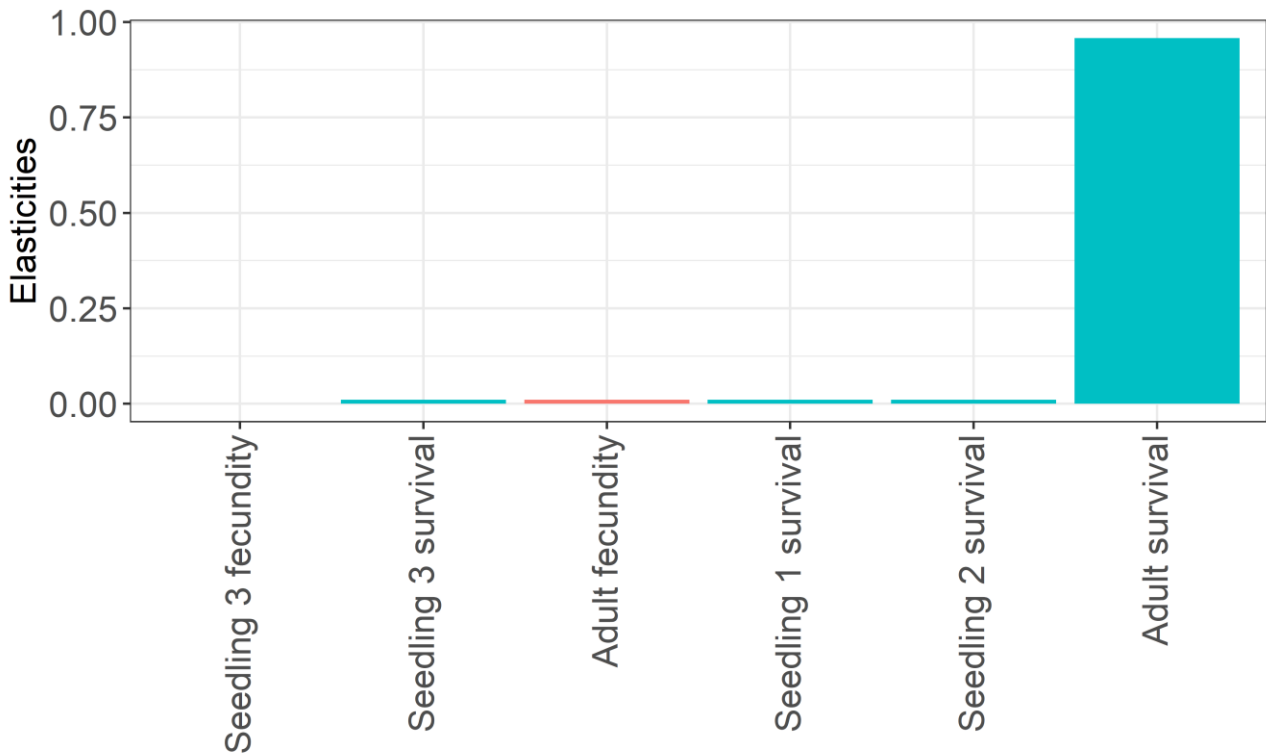
Figure 5.6: Deterministic sensitivity analysis. Elasticities of the stage matrix for the Small golden moth orchid.
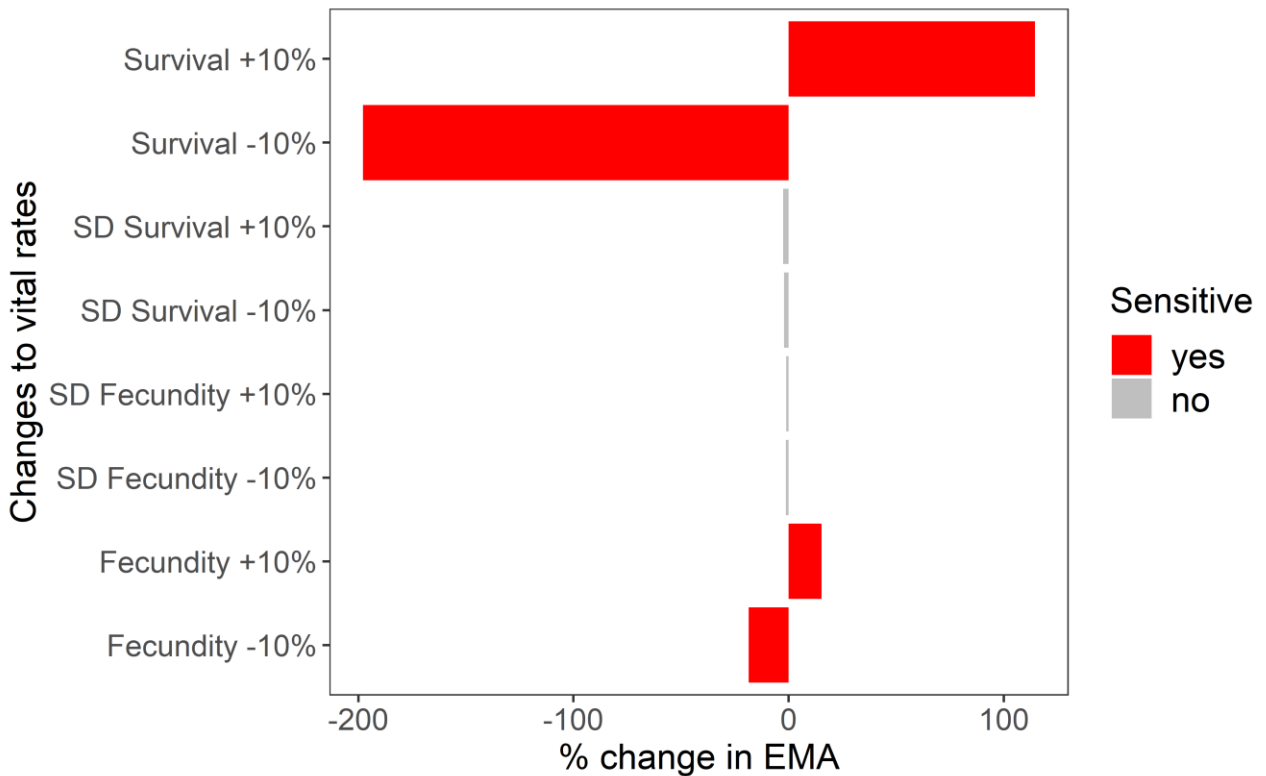
Figure 5.7: Stochastic sensitivity analysis of mean vital rates and standard deviations of the Small golden moth orchid. Changes in EMA greater than 10% (greater than input change) are coloured in red.

Table 5.4: Scenario sensitivity analysis for Small golden moth orchid. EMAs and rank order of management actions (in parentheses). Rank correlations are with base models. Scenario: rabbits and no drought.

Fire scenario	Base models	Constant fecundity	Convex function	Concave survival function	10% reduction in survival
No fire	9 (5)	2 (5)	11 (5)	10 (5)	0 (5)
Short Autumn/Spring	58 (2)	45 (1)	76 (2)	59 (2)	0.4 (1.5)
Medium Autumn	62 (1)	36 (2)	93 (1)	61 (1)	0.4 (1.5)
Medium Spring	41 (4)	30 (3)	61 (4)	41 (4)	0.3 (3)
Long Autumn/Spring	43 (3)	22 (4)	69 (3)	43 (3)	0.2 (4)
Correlation		0.8	1.0	1.0	0.87

5.4   Discussion

A population viability analysis model for the Small Golden Moths Orchid was developed using multiple expert judgements to explore the effectiveness of several management options on the persistence of the species. Results suggest a combined management regime of short to medium fire intervals every 2-3 years in autumn, protecting plants from herbivory from rabbits by installing fences and watering plants during times of drought is needed to help safeguard the species from extinction. The model suggests that rabbits have the greatest negative impact on the species, but there was no one action that was fully effective. Without combining all three actions, the effectiveness of any one action is substantially reduced in the presences of other threats. Before using these results to develop a management plan for the species, consultation with all experts involved in the elicitation will be required to ensure interpretation of model results is correct and recommendations for management are sound.

The sensitivity analysis revealed adult survival is the most important parameter influencing the model. While uncertainty in adult survival did not alter ranks of fire management scenarios, if absolute values of abundance are preferred, then better estimates of survival are necessary. The large influence of adult survival on model behaviour suggests investigating an alternative model structure may be warranted. For instance, including multiple adult stages may provide some extra nuance within the adult stage that may be important for the population dynamics and subsequent management. There is a trade-off as this added complexity will require more information.

The scenario sensitivity analysis investigated several model assumptions for trends in vital rates due to biomass accumulation. This analysis revealed the model is robust to our assumptions, at least for the scenarios tested. It is likely the model will be sensitive to other assumptions that were not tested. In the future it is worthwhile exploring a more in-depth sensitivity analysis to fully explore the behaviour of the model with respect to other important assumptions that were not considered here.

The model is based purely on expert judgement as no empirical data is currently available for the species. To improve our understanding of the species' dynamics and the effectiveness of alternative management options, basic life history data on survival and reproduction under different environmental conditions is needed. For example, very little is known about recruitment and seedling survival for the species. While sensitivity analysis revealed the model was insensitive to these parameters, if experts are completely wrong in their judgement of recruitment and seedling survival, then expectations from proposed management informed by the model may be incorrect. Consultation with species experts and researchers is required to develop a research plan for SGMO on how to best capture the data needed to inform the model including the feasibility of estimating some of the model parameters (i.e. seedling emergence).

One of the largest risks for this species, is the limited number of locations where SGMO is found and therefore a single unforeseen catastrophic event at the remaining population could cause the species to go extinct. This aspect of the species potential fate was not considered in the model development or scenarios. While this study has provided some insights around management of fire, watering and rabbit control, for the

Unofficial

ongoing persistence of the species it will be important to create more populations of SGMO to spread the risk. Creating more populations through translocation of established individuals or through direct seeding should be considered as part of the management plan for the species. The PVA model developed here could be extended and adapted to explore scenarios of population creation and evaluate alternative options for reintroduction.

6 Large-fruit Groundsel (*Senecio macrocarpus*)

6.1 Background

The Large-fruit Groundsel, *Senecio macrocarpus,* is a perennial daisy growing to 70 cm high. It has alternate, linear leaves covered with fine hairs on both surfaces. It employs an opportunistic growth mechanism – it may remain green all year round, or die back in dry seasons then re-sprout from rootstock depending on rainfall and fire regime (Cutten and Squire 2003). Seed production and germination rates are highly variable between seasons. In years with sustained spring and summer rainfall, the plant may produce many flushes of flower heads from September to November, and occasionally in March and April (Belcher 1983, Walsh 1999), while in dry seasons growth and flowering are substantially reduced.

The Large-fruit Groundsel (LFG) occurs in grasslands, sedgelands, shrublands and woodlands, generally on sparsely vegetated sites on sandy loam to heavy clay soils, often in depressions that are waterlogged in winter. In Victoria, the species has been recorded widely across the State, from near Horsham in the west to near Omeo in the east, with most records from western Victoria (Hills and Boekel 1996).

There has been a substantial decline in range and abundance of the LFG. Current major threats include ongoing disturbance and destruction of habitat, competition, weed invasion and possibly impacts from climate change (Fordham et al. 2012). The initial cause of decline of *S. macrocarpus* was the spread of agriculture across the grassy plains of south-eastern Australia. The remaining populations survive mostly in remnant habitats along rail lines and in small reserves (Sinclair 2010). including two natural populations within the MSA area (DELWP 2015).

We adapted an existing PVA model for LFG that was developed to explore impacts of climate change on plant persistence (Fordham et al. 2012)

6.2 Methods

6.2.1 Conceptual models

The conceptual model was developed that outlines the LFG's ecology and key life history stages, the processes and drivers that influence the plant's survival at each stage, and the management actions that can be used to ameliorate these threats (Figure 6.1). The species has been described in three main stages, seedbank, juveniles (age 0-2) and established adults (age 3-8). Plants become reproductive above two years of age. Seeds are thought to persist in soil more than a year, and hence we incorporated a seedbank stage. This simplified conceptual model informed the development of the management population models for the species.

The main threats to LFG within the MSA management area are competitive biomass, prolonged drought, sustained grazing from livestock and inappropriate burning regimes. The accumulations of biomass over time is thought to reduce the survival of all plants but at different stages depending on juvenile or adult life stages (Figure 6.3). Prolonged drought increases mortality of plants.

The main management levers include fire as a biomass control agent to reduce threats of biomass accumulation and watering plants during times of drought. We assume that grazing of stock will be removed where LFG populations exist under MSA management.

Unofficial
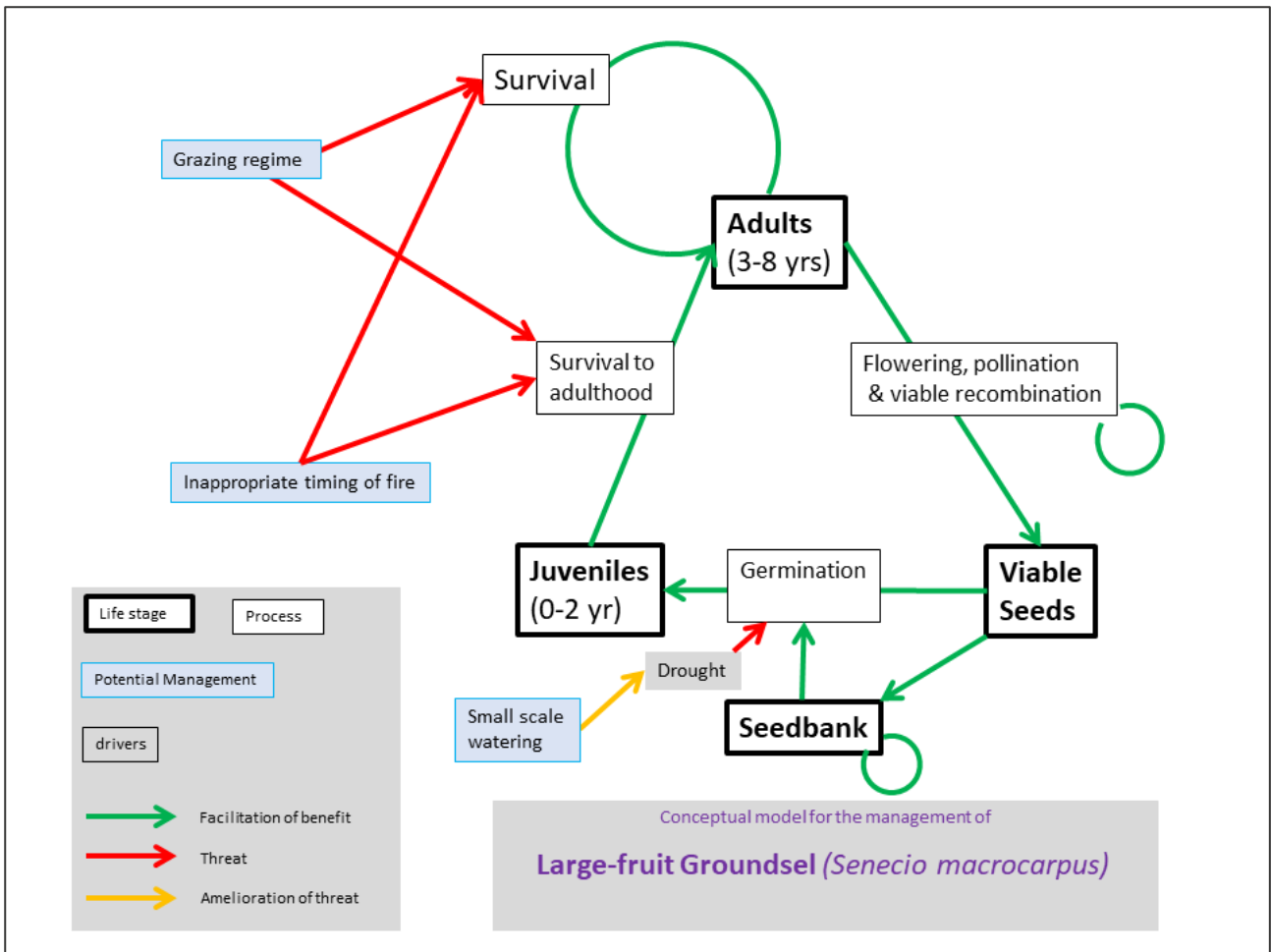
Figure 6.1: Conceptual mode of the life cycle of Large-fruit Groundsel including how threats and management influence different stages of the life cycle.

The timeline of important reproductive events and responses to fire seasons (Figure 6.2) provides context for the way burning can ameliorate the threats of biomass accumulation. There is some disagreement between experts regarding the species' needs for open ground for seed germination and seedling establishment. For the model we assume low biomass is beneficial for seed germination and establishment. While adults can persist for years in closed swards of grass, we assume seedlings cannot establish, and local populations may eventually die out without burning to reduce biomass accumulation. Burning has positive and negative effects on survival. Adult plants can re-sprout from rootstock after fire (Cutten and Squire 2003). Spring fire removes reproductive outputs for the year if it occurs at inopportune times in the growing season, particularly when the first shoots are emerging from rootstock. In the short-term there may be immediate mortality from fire but in the long-term the survival rates are higher for those plants that survive fire or germinate after a fire due to less competition (Figure 6.1).
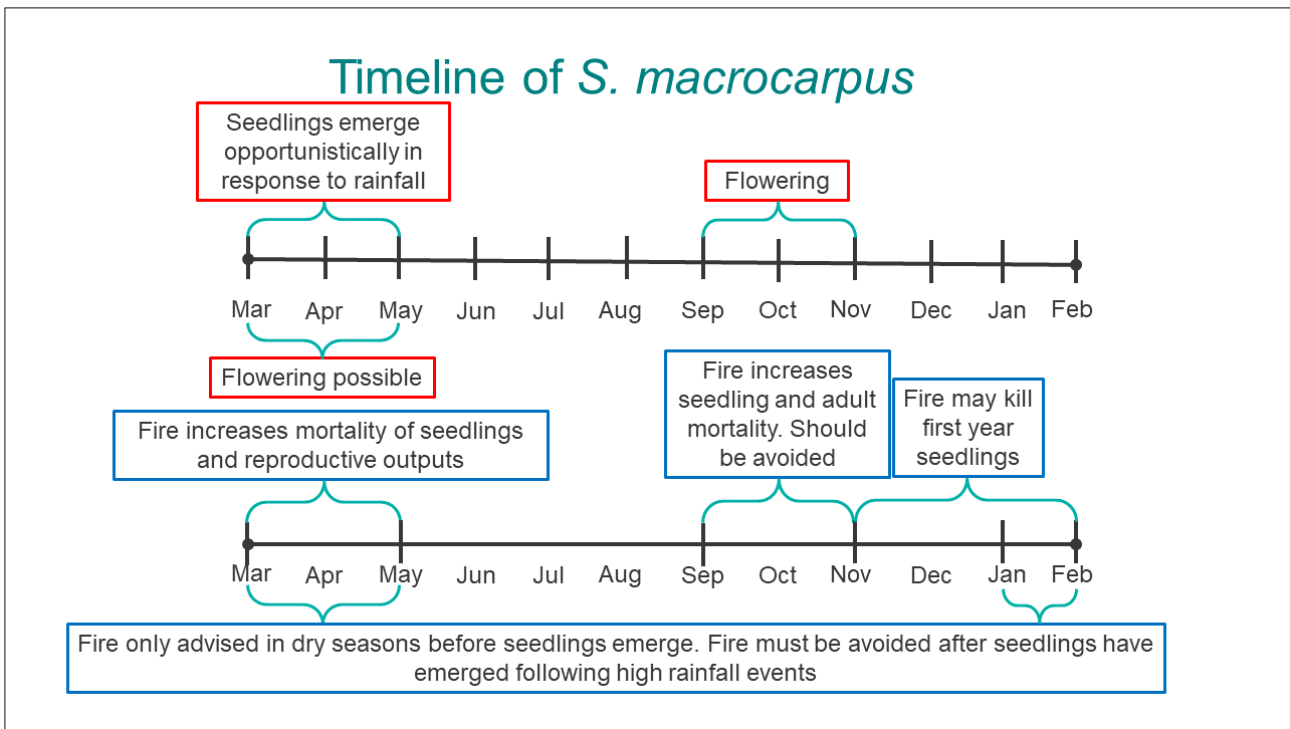
Figure 6.2: Conceptual mode of the timing of ecological processes and effects of biomass control in different seasons.

6.2.2 Vital rates

We adapted an existing PVA model for LFG that consisted of 12 stages (Fordham et al. 2012). We modelled LFG using 10 age classes, a seedbank, three juvenile age classes and six adult age classes reflected a lifespan of 8-9 years.

Survival rates

Parameter estimations were derived from an empirical study and published expert judgments used to populate a previous PVA model (Table 6.1). Survival rates in the base model (grassland biomass accumulates, no biomass control, rainfall pulse 1 in 8 years) were estimated for ten stage matrix components; seedbank, juvenile 1, juvenile 2, juvenile 3, adult 1, adult 2, adult 3, adult 4, adult 5, and adult 6. Values for germination from seed, juvenile 1 and juvenile 2 survival were taken from (Zamin et al. 2018). Survival rates of the remaining stages were taken from (Fordham et al. 2012)

Fecundity

Effective fecundity rate was calculated by combining multiple population parameters that were elicited and estimated separately from expert workshops and from Fordham et al. (2012). (i.e. the number of inflorescences per plant; percentage of inflorescences aborted per plant, and number of seeds produced per inflorescence were combined to calculate the number of seeds produced per plant) (Table 6.1).

Table 6.1: Life stage parameters, values and description for Large-fruit Groundsel.

Parameter	Value	Description
Adult 1 fecundity	7	Annual number of viable seeds per plant
Adult 2 fecundity	14	Annual number of viable seeds per plant
Adult 3 – 6 fecundity	20	Annual number of viable seeds per plant
Seedbank survival	0.02	Annual proportion of seeds surviving each year
Seedbank to Juv 1	0.04	Annual proportion of seeds germinating each year and surviving to the census

Unofficial

Juv 1 to Juv 2	0.37	Annual proportion of 1st year juveniles surviving to their 2nd year
Juv 2 to Juv 3	0.36	Annual proportion of 2nd year juveniles surviving to their 3rd year
Juv 3 to Adult 1	0.5	Annual proportion of 3rd year juveniles surviving to adulthood
Adult 1 to Adult 2	0.7	Annual proportion of stage 1 adults surviving transitioning to stage 2 adults
Adult 2 to Adult 3	0.7	Annual proportion of stage 2 adults surviving transitioning to stage 3 adults
Adult 3 to Adult 4	0.4	Annual proportion of stage 3 adults surviving transitioning to stage 4 adults
Adult 4 to Adult 5	0.4	Annual proportion of stage 4 adults surviving transitioning to stage 5 adults
Adult 5 to Adult 6	0.1	Annual proportion of stage 5 adults surviving transitioning to stage 6 adults
Adult 6	0.0	Annual proportion of stage 6 adults surviving

6.2.3 Stochasticity

Demographic stochasticity was incorporated in the model to reflect the level of natural variation in survival and fecundity rates. This variation was unknown and was not elicited, therefore we used default values that were assumed for all the plant species PVA models where standard deviations could not be estimated from the data. For all vital rates (i.e. fecundities, survival of recruits and juveniles and survival of adults), standard deviations were derived from a co-efficient of variation of 20% of the mean for each vital rate. This assumption was evaluated as part of the sensitivity analysis. Once more empirical data on variability in vital rates becomes available, updated estimates for the standard deviations of vital rates can be incorporated into the model.

6.2.4 Biomass accumulation

We assumed biomass accumulation has an adverse effect on LFG populations due to reduced space for germinants to establish and increased competition. Biomass accumulation was modelled as a temporal trend that relates to the time since biomass control on x-axis as a function of annual survival rate of the species (Figure 6.3). The shape of the function between low (1 year since biomass control) and high biomass (5+ years since biomass control) was not directly elicited from expert workshops. Subsequent consultation with grassland experts suggested that the function was non-linear and resembled an inverted logistic function where the decline to survival rate accelerates after three years since biomass control. A linear and additional non-linear relationship between survival rate and TSBC were investigates as part of the sensitivity analysis (Figure 6.4).
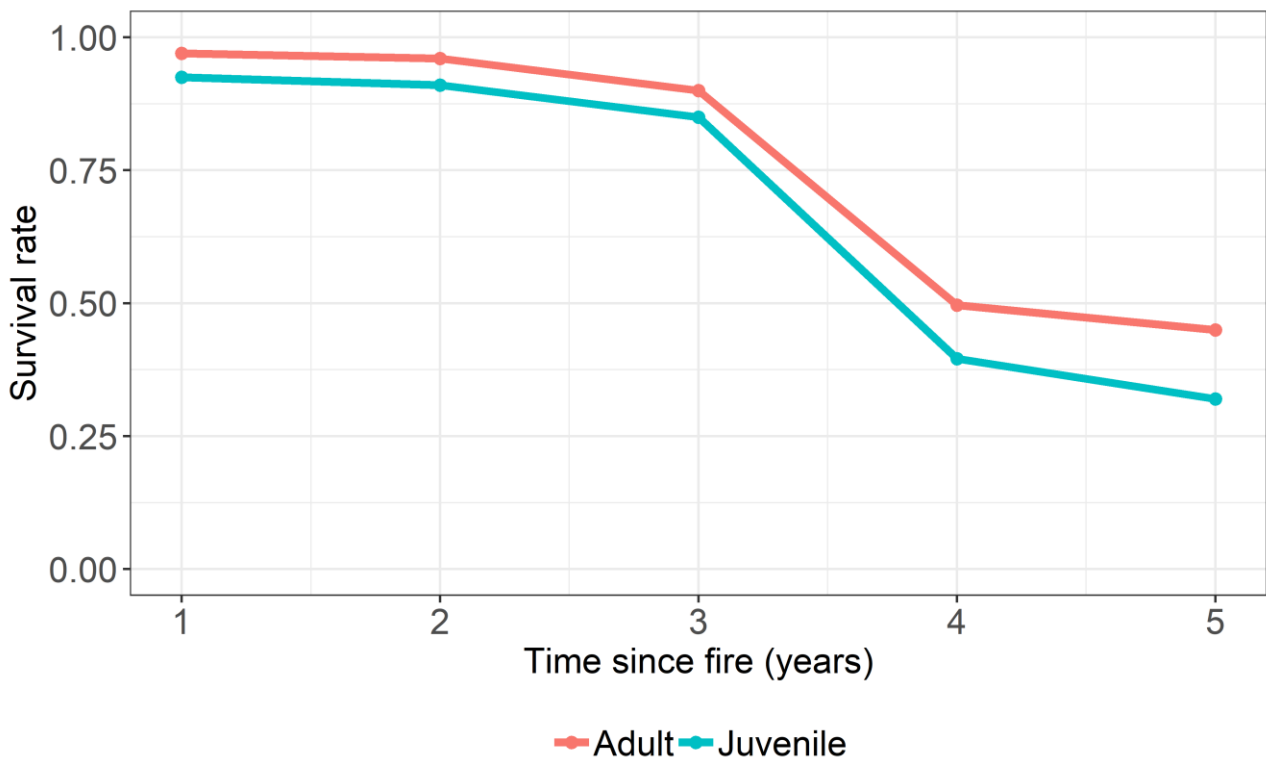
Figure 6.3: Relative change in survival rate of Large-fruit Groundsel in relation to time since biomass control (TSBC).

6.2.5 Additional model specification

Density dependence

A ceiling density dependence function is assumed across all stages. This means that modelled populations can grow exponentially until they reach a limit (ceiling), and then remain at that level. The carrying capacity was set to 10 million, a level too high for ceiling capacity to be reached by the model's timespan. The population growth was regulated through relationship between biomass accumulation (when increased, reduces growth rate) and controlled burning (when increased, increases growth rate).

Initial conditions

The initial population size was set at 1415 and we assumed a stable age distribution. This resulted in the following stage specific initial abundances of: seedbank= 846; juvenile stage 1 = 338; juvenile stage 2 = 125; juvenile stage 3 = 45; adult stage 1 = 23; adult stage 2 = 16; adult stage 3 = 16; adult stage 4 = 11; adult stage 5 = 8; adult stage 6 = 0. The initial abundance in the model was set much larger than at known populations to allow exploration of the dynamics of the model and to explore a range of model scenarios. Setting the initial abundance too low can cause extinction under all scenarios and would not allow for a valid comparison.

The model was simulated 1000 times over 50 time-steps (1 time-step = 1 year). Model scenarios were compared using expected minimum abundance (EMA), the average of the minimum values for each trajectory of a stochastic populations model.

6.2.6 Management scenarios

Eight management scenarios were explored using eight individual models (Table 6.2), including the null or "no management" scenario. Fire as biomass control and watering to promote germination were the two main types of management actions considered in the models. Elicited values of immediate mortality and reduced fecundity due to types of biomass control (fire interval) were incorporated into each of the scenarios. Watering to increase germination was assumed to happen one in six years (c.f. one in eight in the "no watering").

Table 6.2: Summary of management scenarios considered in each model.

Scenario number	Description	Fire interval	Fire season	Watering regime
1	No fire management, no watering	No fire management	NA	no watering
2	Short fire interval, Autumn/Spring, no watering	Short	Autumn/Spring	no watering
3	Medium fire interval, Autumn/Spring, no watering	Medium	Autumn/Spring	no watering
4	Long fire interval, Autumn/Spring, no watering	Long	Autumn/Spring	no watering
5	No fire management watering	No fire management	NA	watering
6	Short fire interval, Autumn/Spring, watering	Short	Autumn/Spring	watering
7	Medium fire interval, Autumn/Spring, watering	Medium	Autumn/Spring	watering
8	Long fire interval, Autumn/Spring, watering	Long	Autumn/Spring	watering

6.2.7   Sensitivity analysis

Three types of sensitivity analysis were investigated: deterministic, stochastic and scenario sensitivity. The deterministic sensitivity analysis focused on the elasticities of the stage matrix, i.e. the proportional contribution of each of the parameter estimates toward the growth rate implied by the stage matrix. The stochastic sensitivity analysis involved adjusting each of the vital rates and standard deviations of a stage matrix separately by ±10% to investigate if the stochastic model was sensitive to any these parameters. If the EMA results changed by more than ±10% this indicates the model is sensitive to that parameter.

The scenario sensitivity analysis evaluated two key assumptions in the model to determine whether the rank order of fire management scenarios change compared the base scenarios. In all base scenarios, the temporal trend in survival rates due to biomass accumulation was assumed to be an inverted logistic curve relationship (see Figure 6.3). A linear and an exponential decay function were investigated (Figure 6.4) to test the effect of biomass accumulation on survival rates. Secondly sensitive parameters from the stochastic sensitivity (i.e. survival rates and fecundity) was investigated to see whether that sensitivity resulted in a change in the rank order of fire management actions.
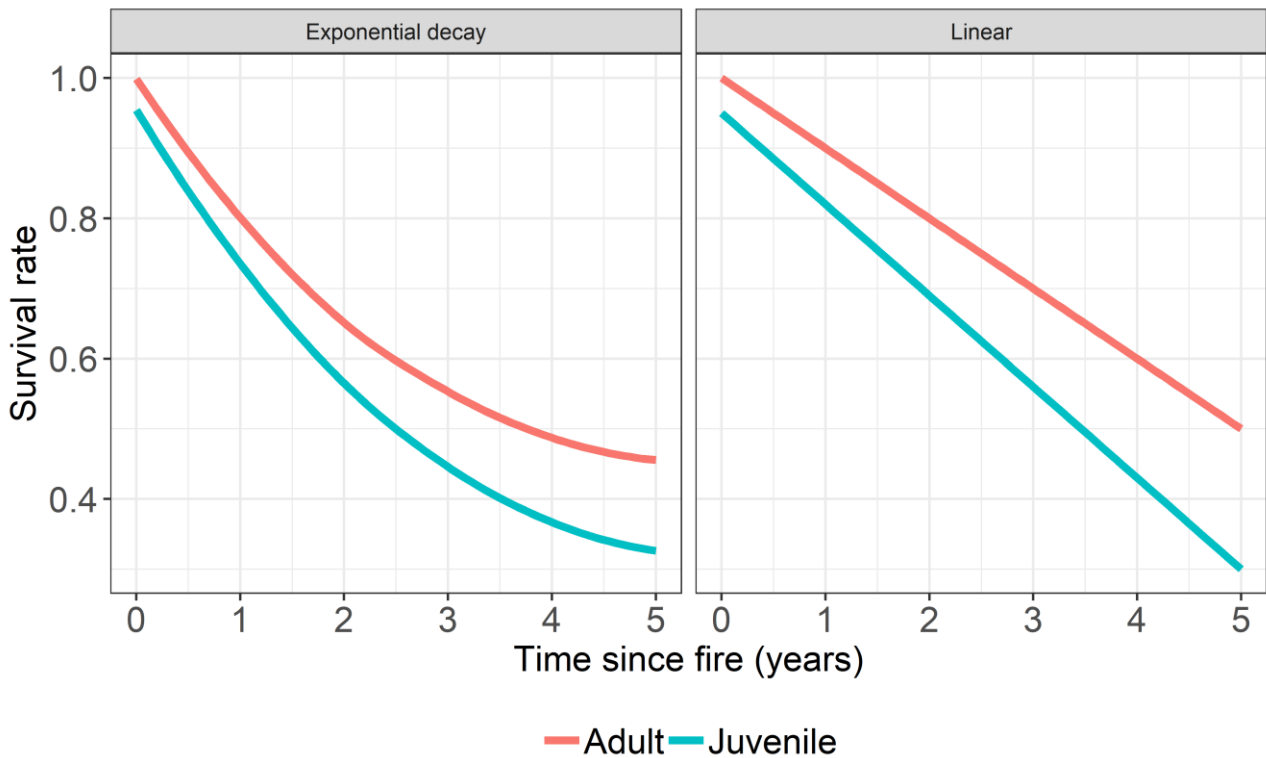
Figure 6.4: Functional forms tested for trend in survival rates given TSBC used in scenario sensitivity analysis.

6.3   Results

6.3.1   Management actions

Results suggest short fire intervals to control biomass accumulation combined with watering to promote germination have the best outcomes for LFG. Short fire intervals without watering is the second preferred option suggesting biomass control brings larger benefits to the species compared to watering. Long fire intervals and no biomass control seem to be detrimental to LFG irrespective of watering or not (Figure 6.5).

Unofficial
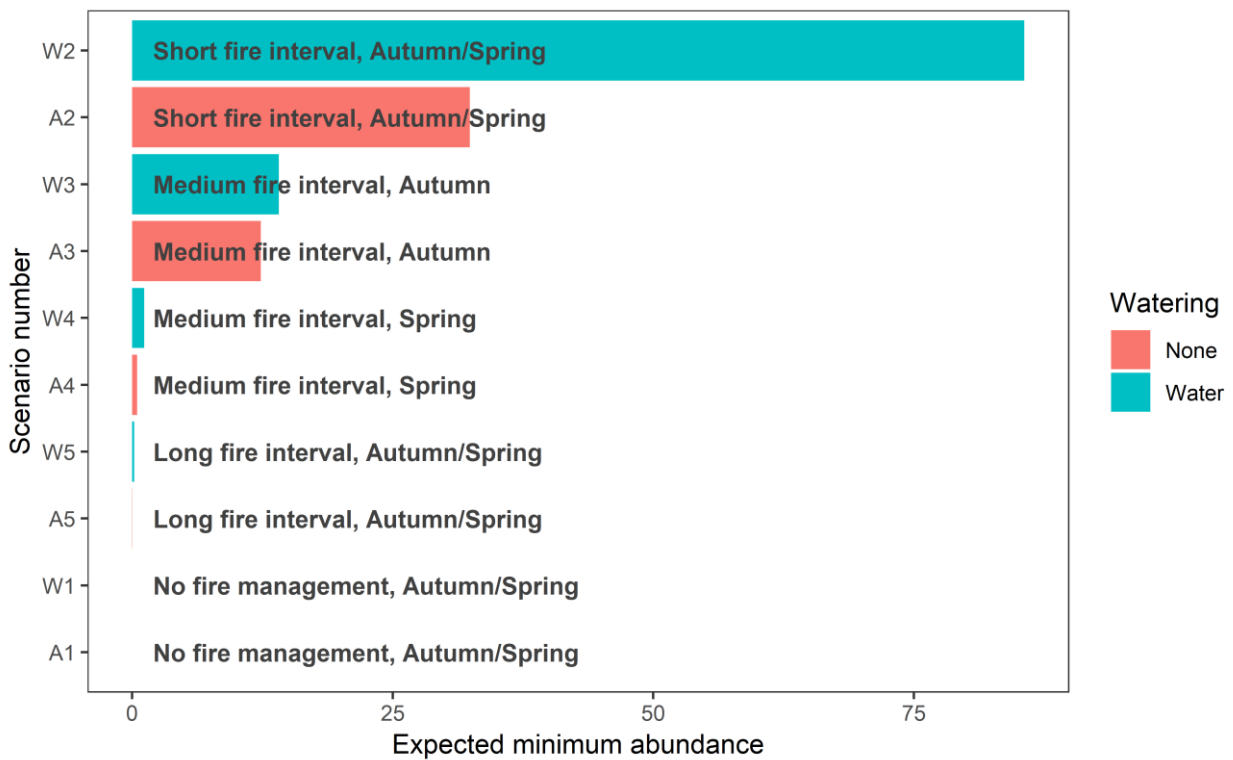
Figure 6.5: The result of PVA for Large-fruit Groundsel. Ranking of management actions and subsequent EMA for each scenario.

6.3.2 Sensitivity analysis

Deterministic sensitivity analysis

Deterministic sensitivity analysis revealed that the model was most sensitive to juvenile l and young adult (adult 1 and 2) survival and moderately sensitive to fecundity, with the exception of older adult stages (Figure 6.6).
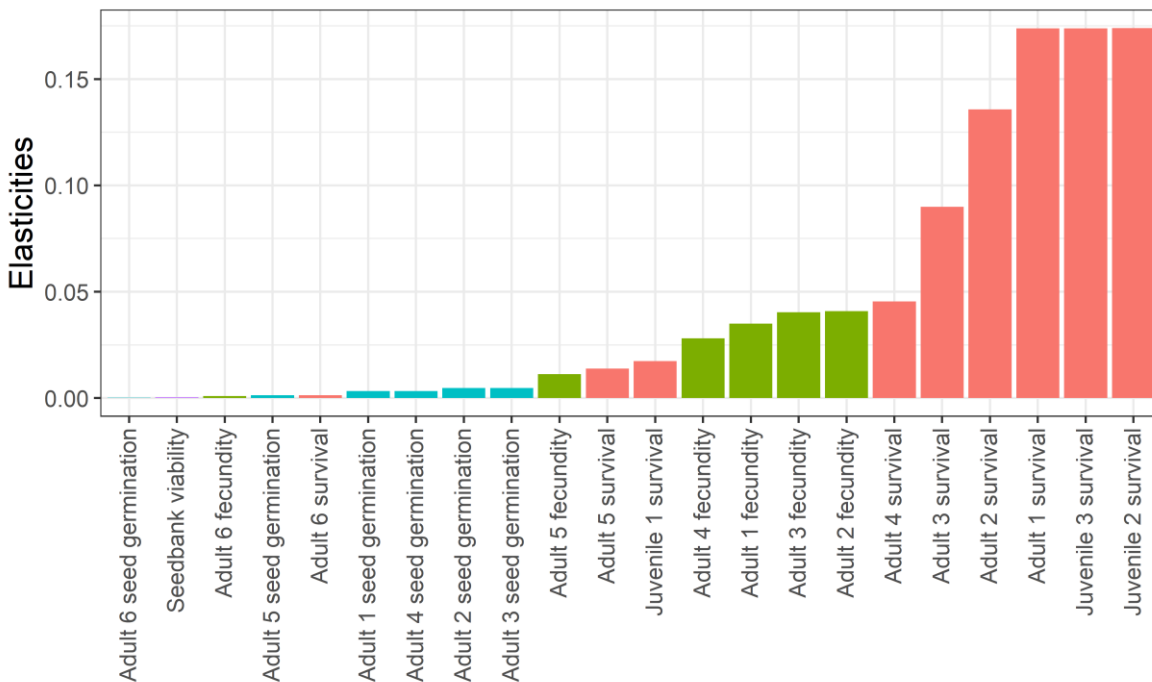
Figure 6.6: Result of deterministic sensitivity analysis. Elasticities of each life stage matrix parameter ranked in ascending order.

Stochastic sensitivity analysis

A model is sensitive to an input parameter if certain percentage change in the parameter results in a larger percentage change in the output of interest (i.e. expected minimum abundance). We tested the impact of +/- 10% changes in survival rate, fecundity and standard deviation matrix of both rates. Comparisons were based on the EMA of model #6, or the "short fire, watering" scenario.

Stochastic sensitivity analysis revealed that the model is most sensitive to changes in survival rate. The model is also sensitive to changes in fecundity and the standard deviation of survival (Figure 6.7).
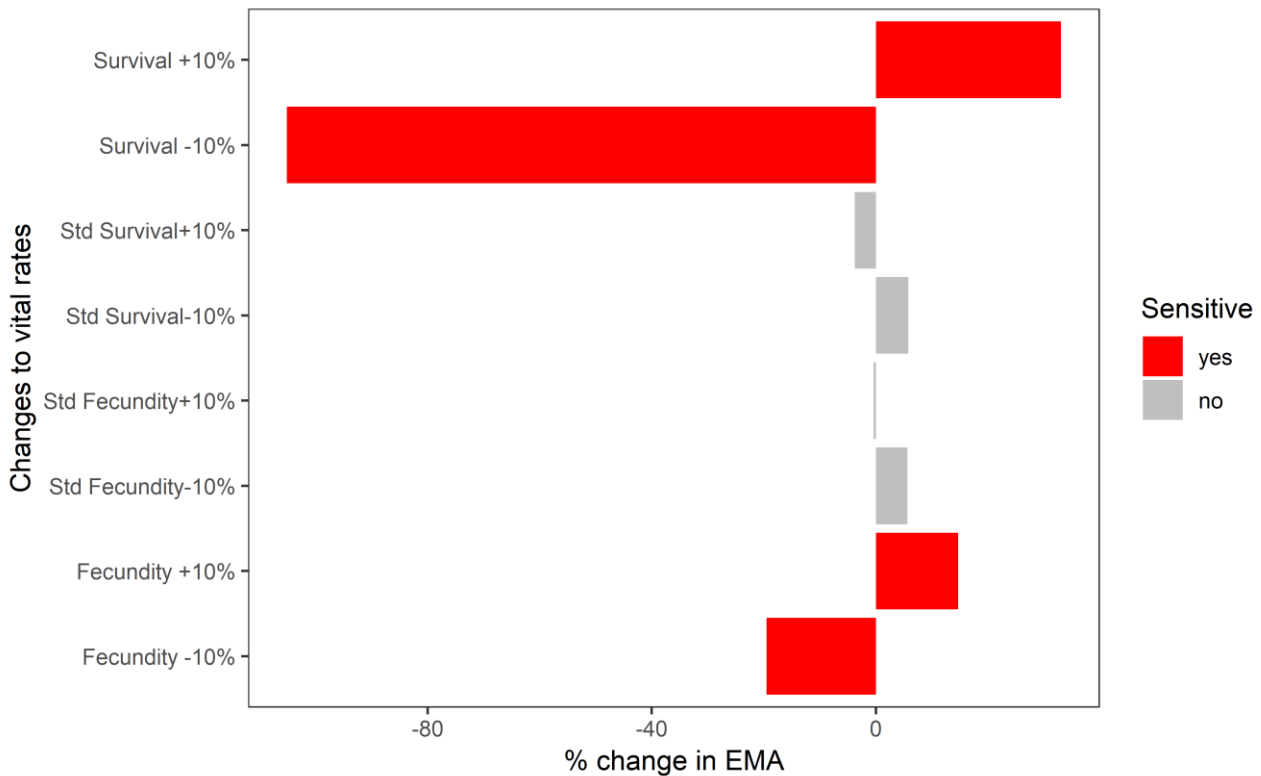
Figure 6.7: Result of stochastic sensitivity analysis. Changes in EMA greater than 10% (greater than input change) are coloured in red.

Scenario based sensitivity analysis

The outcomes of the scenario sensitivity analysis suggested that the ranks of actions are not sensitive to changes in survival rate or fecundity (i.e. correlation coefficients are 0.99 and 0.98 respectively). We tested the impact of linear and exponential decay relationships between time since biomass control and survival rate to explore the potential alternative outcomes on the rank order of the management actions. An exponential decay function resulted in the same ranks of management actions as the base model (Table 6.3). When a linear function was used to represent change in survival rates given time since biomass control, there was a slight change in the rank order of management actions and a correlation co-efficient of 0.75. However, the most preferred action, short fire interval with watering did not change rank order (Table 6.3).

The result of this investigation suggests that "short fire and watering" scenario (model #6) returns the highest EMA across all three types of assumptions tested. All four assumptions tested suggest that "no fire" with or without watering (models 1&5) are the worst management action in terms of expected minimum abundance for LFG.

Table 6.3: Scenario sensitivity analysis for LFG. EMAs (in parentheses) and rank order of management actions. Rank correlations are with base models. *Indicates tied ranks (EMA = 0).

Scenario number	Base model	Linear function	Exponential decay function	When survival is reduced by 10%	When fecundity is reduced by 10%
	EMA (Rank)	EMA (Rank)	EMA (Rank)	EMA (Rank)	EMA (Rank)
A1	0 (9.5)	0 (9.5)	0 (9.5)	0 (9.5)	0 (9.5)
A2	16.203 (2)	0.714 (2)	0.024 (2)	0.513 (2)	19.692 (2)
A3	6.174 (4)	0.043 (5)	0.001 (3)	0.297 (4)	3.645 (4)
A4	0.233 (6)	0.006 (6)	0 (9.5)	0.003 (6.5)	0.269 (6)
A5	0.014 (8)	0 (9.5)	0 (9.5)	0 (9.5)	0.022 (8)
W1	0 (9.5)	0 (9.5)	0 (9.5)	0 (9.5)	0 (9.5)
W2	85.603 (1)	9.685 (1)	0.12 (1)	4.227 (1)	69.759 (1)
W3	14.084 (3)	0.408 (3)	0 (9.5)	0.439 (3)	12.452 (3)
W4	1.157 (5)	0.101 (4)	0 (9.5)	0.026 (5)	0.821 (5)
W5	0.182 (7)	0.001 (7)	0 (9.5)	0.003 (6.5)	0.043 (7)
Correlation coefficient		**0.75**	**1**	**0.99**	**0.98**

6.4 Discussion

We developed a population viability analysis model for LFG using multiple expert judgements, data from a germination trial and values in a previously published PVA to explore the effectiveness of several management options on the persistence of the species. Results suggest a combined management regime of short fire intervals (every 1-2 years) in late Spring or early Autumn (prior to the Autumn break) and watering to stimulate germination is needed to help safeguard the species from extinction. Without combining both actions, the effectiveness of any one action is substantially reduced. Before using these results to develop a management plan for the species, consultation with all experts involved in the elicitation will be required to ensure interpretation of model results is correct and recommendations for management are sound.

The deterministic sensitivity analysis revealed that juvenile survival is an important parameter influencing the model, whist stochastic sensitivity revealed that survival and fecundity were influential. While neither uncertainty in survival or fecundity altered ranks of fire management scenarios, if absolute values of abundance are preferred, then better estimates of these parameters are necessary. The large influence of juvenile survival and fecundity on model behaviour suggests investigating an alternative model structure may be warranted. For instance, a better understanding of the link between seed production, viability and germination and the factors influencing juvenile mortality such as the direct impacts of fire will improve model accuracy.

The scenario sensitivity analysis investigated two model assumptions for trends in vital rates due to biomass accumulation. This analysis revealed the model is generally robust to our assumptions, at least for the scenarios tested. Changing the shape of the biomass/survival relationship did not change the ranks of fire scenarios. We were however unable to rank some alternative scenarios due to them having an EMA of zero. This may be due to the relatively small population modelled in the MSA area. It is likely the model will be sensitive to other assumptions that were not tested. In the future it is worthwhile exploring a more detailed sensitivity analysis to fully explore the behaviour of the model with respect to other important assumptions that were not considered here as well as any differing opinions among experts.

The model is based on a mixture of expert judgements, empirical data and values in a previously published PVA. As empirical data is only available for juvenile life stages improving our understanding of the species dynamics how it responds to alternative management options, basic life history data on survival and reproduction under different environmental conditions is needed. Furthermore, information on how the season of fire influences reproduction and survival may improve the model, particularly the seasonal impacts of fire. Consultation with species experts and researchers is required to develop a research plan for LFG on how to best capture the data needed to inform the model.

One of the largest risks for this species is the small population size as a single unforeseen catastrophic event could cause the population to go locally extinct. This aspect of the species potential fate was not considered in the model development or scenarios. While this study has provided some insights around management of fire and watering, for the ongoing persistence of the species it will be important to create more populations of LFG and or to increase the size of existing populations to spread the risk. Creating or enhancing populations through translocation or reintroduction of established individuals or through direct seeding should be considered as part of the management plan for the species. Reintroduction has resulted in successful establishment and recruitment (Morgan 1999b). The PVA model developed here could be extended and adapted to explore scenarios of population creation and evaluate option for how it can be achieved.

Unofficial

7 Golden Sun Moth (*Synemon plana*)

7.1 Background

The Golden Sun Moth (*Synemon plana*) typically occurs in native grassland and grassy woodland dominated by wallaby-grass (DSE 2004). Prior to European settlement, Golden Sun Moth (GSM) was widespread and relatively continuous across south-eastern Australia's grasslands but loss of habitat has resulted in fragmentation of the habitat and isolation of populations. GSM has a complex, relatively unknown life history with four main life-stages; eggs, larvae, pupae and adult. During its early life stages, GSM resides underground or deep inside tussocks where adequate forage is available. The short-lived adult moths emerge between mid-October and early January, and are active in the middle of warm, sunny and relatively still days. Adult males fly constantly over the grassland in search of the relatively immobile females, which sit on the ground and display their golden hindwings. The mated females lay their eggs around the bases of tussock grass. (Department of the Environment 2012).

Major threats to the persistence of the species, particularly in the MSA management area mostly relate to habitat loss, resource availability (e.g. food and shelter), disturbance (e.g. grazing, fire and predation by birds) and fragmentation due to agricultural expansion and urbanisation (e.g. weed invasion and over fertilization). Many populations are isolated due to highly fragmented habitats, impeding the ability of the relatively immobile females to recolonise areas, thereby reducing the likelihood of genetic exchange (DSE 2004).

7.2 Methods

7.2.1 Conceptual models

The conceptual model outlines the GSM's ecology and key life history stages, the processes that influence the moth's survival at each stage, and the management actions that can be used to ameliorate these threats and increase moth survival (Figure 7.1). This conceptual model and the associated annual timeline (Figure 7.2) informed the development of the population viability models for the species.
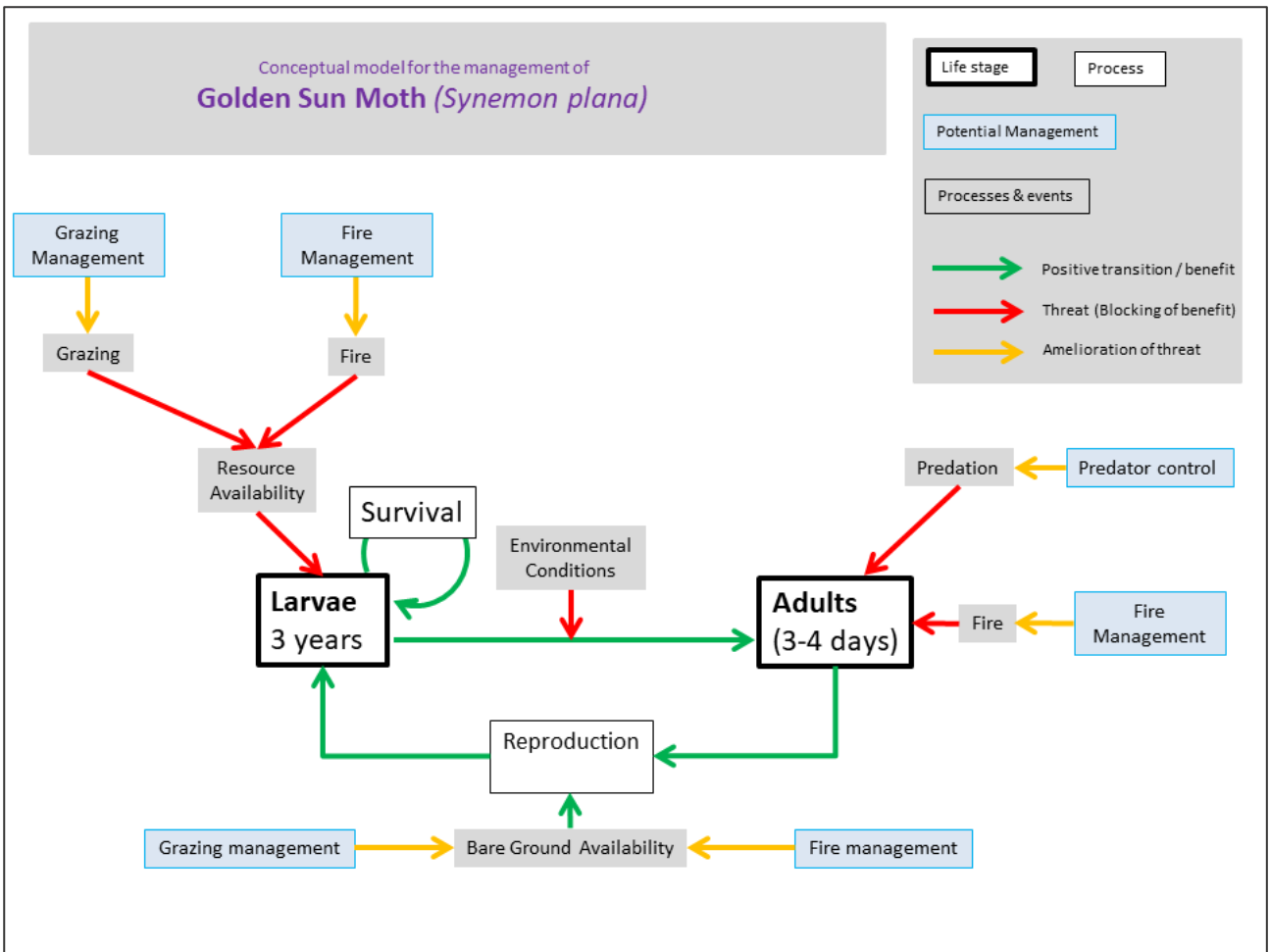
Figure 7.1: Conceptual model for the Golden Sun Moth.
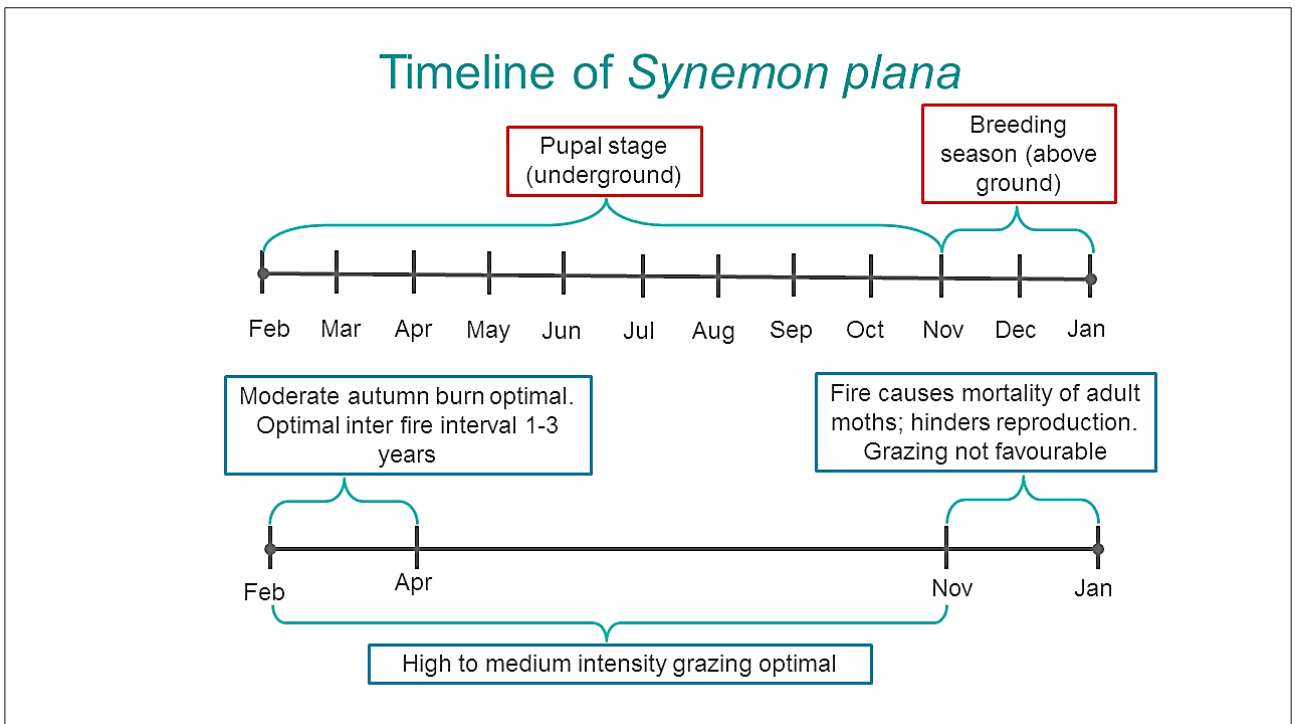
Figure 7.2: Annual lifecycle of Golden Sun Moth and the effects of biomass control.

Unofficial

7.2.2 Vital rates

Survival rates

The model assumes two life stages; larvae and adult. The parameters for larval lifespan, survival and emergence (transition from larvae to adult) were derived from expert elicitation. Larvae were assumed to live in this stage for up to three years, with individuals able to emerge as adults in years two and three. The model assumed adults live for three to four days and thus have an annual survival rate of 0.

The population growth rate was estimated from population monitoring data between 1993 and 2006 by (Richter et al. 2013). The average annual population growth rate over this period was calculated to be 1.018.

Fecundity

Fecundity is related to the amount of biomass, this is explained below in section 7.2.4.

7.2.3 Stochasticity

Both demographic and environmental stochasticity are included in the model. The stage matrix for GSM is shown in Table 7.1.

Table 7.1: Life stage parameters for Golden Sun Moth.

Coefficient of variation set to 10% for fecundity and survival.

Parameter	Value	Description
Adult fecundity	1.48	Annual number of eggs that hatch as larvae per adult
Larval survival	0.5	Annual proportion of larvae surviving
Laval emergence	0.8	Annual proportion of larvae emerging as adults
Adult survival	0.0	Annual proportion of adults surviving

7.2.4 Biomass Accumulation

Biomass accumulation is thought to have adverse effects on GSM population growth. Direct effects of biomass accumulation in the PVA were considered on two independent parameters; fecundity and carrying capacity of the habitat that is suitable for golden sun moths.

Expert consultation indicated that the proportion of moths successfully mating is directly related to the ability of males to locate mates in amongst the grass, which is negatively affected by the amount of biomass i.e. the proportion of males able to find a mate is lower when biomass is higher. Therefore, GSM fecundity is assumed to be inversely correlated with biomass accumulation. The model assumes a constant rate of biomass accumulation with time since biomass control (see Figure 1.1). To determine the relationship between time since biomass reduction and the amount of biomass we used data collected during the annual MSA grassland monitoring (DELWP 2015). Biomass as estimated by the golf ball visibility score (Schultz et al. 2018). We then combined the linear relationships between the amount of biomass and time since fire and with expert estimations of the percentage of moths mating at different levels of biomass to model the relationship between time since biomass control (fire or grazing) and the percentage of moths successfully mating (Figure 7.3).
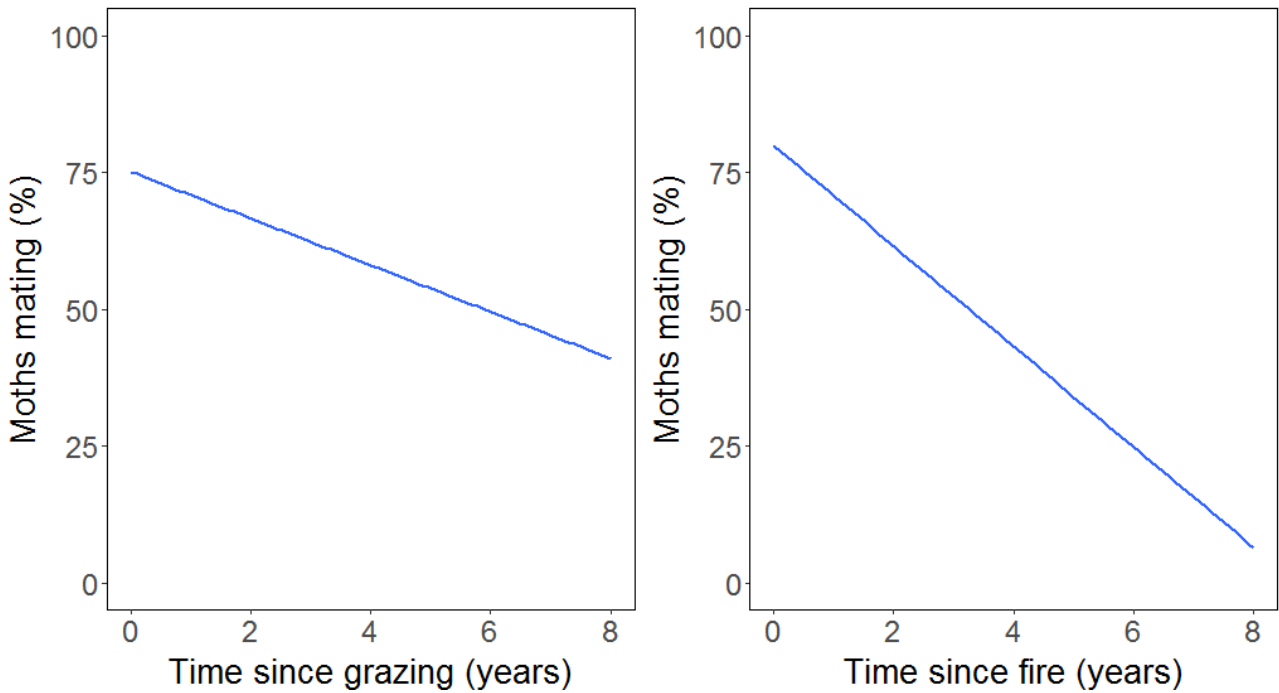
Figure 7.3 Relationships used to determine the percentage of moths successfully mating given for time since grazing and time since fire.

The example shown for grazing assumes moderate intensity (2.3 DSE ha^{-1}). The average golf ball score for each management unit was calculated with a known recent biomass reduction event (fire or grazing). We assume a linear relationship between the time since the last biomass reduction event and the amount of biomass. These relationships were different for grazing and fire (Figure 1.1).

Habitat suitability for GSM correlated with cover of food plants, specifically C3 (cool season active) grasses (Kutt et al. 2016). From field data, we established a positive relationship between time since fire and the average cover of C3 grasses (native and exotic) in a management unit in the MSA area (Figure 7.4). The slope of this relationship was used to increase the carrying capacity in the model as time since fire increases. There was no apparent relationship in that data between time since grazing and the cover of C3 grasses, so carrying capacity was not adjusted under grazing regimes or under no biomass control.
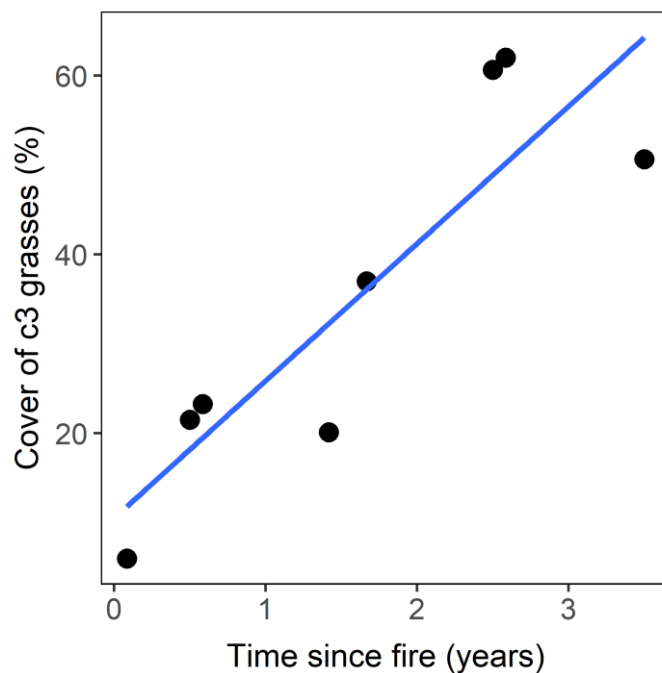
Figure 7.4: Relationship between C3 grass accumulation and time since fire (months).

Unofficial

7.2.5 Additional model specifications

Density dependence

Model assumes a ceiling density dependence function. We modelled two different patch sizes, large and small reserves, because GSM is found in both the Western Grasslands Reserve and smaller isolated conservation reserves and the degree of threats and possible management actions vary in the different reserve sizes. Carrying capacities were calculated for large and small reserve based on the average area of four land management units currently within the Western Grassland Reserve (223 ha) and the average area of the seven GSM reserves (70 ha) respectively. For large reserves, the initial carrying capacity was set at 1.5 million and for small reserves it was approximately one sixth of the large reserve value, these values are based on density estimates published in (Gibson and New 2007).

7.2.6 Management scenarios

Biomass reduction scenarios for GSM habitats comprise of two types of biomass reduction events; fire and grazing. These events were modelled separately.

Biomass control - Fire

Planned fire was modelled as a reverse catastrophe (i.e. with presumed positive benefit). We model planned fire probabilistically to account for practical and feasibility issues that may prevent a planned fire from occurring in a specific year. Three fire return interval scenarios were considered in the model; short, medium and long (median intervals of 2, 3 or 5 years).

Planned fires in grasslands are usually applied in spring or autumn. Adult GSM are active in spring and it is thought they will be directly impacted by a fire. We incorporated the following assumptions into the scenarios with spring fire.

- Moths flying on the day of the fire are killed and thus there are no mating opportunities on that day

- Moths typically fly on sunny days. There are 44 flying days per year, based on the average number of cloud-free days from mid-October to the first week in January (data from Australian Bureau of Meteorology).

- If a fire occurs in a year then the percentage of moths mating in that year is lowered by 2.3% (i.e. 1 day in 44 flying days.

Due to limitations in the modelling software, burning in different seasons was not possible within the same simulation. For model scenarios that included both Autumn and Spring fires, the effect on the percentage of moths mating was adjusted to account for the different proportions of spring and autumn fires occurring across the 50-year time frame (Table 1.2)

Biomass control - Grazing

We modelled three levels of grazing intensity; low, medium and high (Table 7.2). These levels are approximately equivalent to 1.5, 2.3 and 5.6 DSE ha^{-1}. We assumed the slope of the relationship between grazing and biomass is constant (the rate of biomass accumulation) and that grazing intensity modifies the intercept (the amount of biomass at time 0). We assumed that our observed relationship was from medium grazing intensity and we modified the biomass reduction according to the effect sizes calculated by (Eldridge et al. 2016). The intercept for high grazing intensity was set at a golf ball score of 18 (minimum measurable biomass). Grazing is assumed to occur annually with high intensity grazing occurring as a pulse of less than four weeks, while medium and low intensity grazing occurring over two or three months. It is assumed that grazing does not occur at the same time of year for more than 2 or 3 years. Grazing is not considered feasible in small reserves and therefore was not considered as a management action in our modelled scenarios for small reserves. Large reserves have a grazing scenario.

Table 7.2. Adjustment percentage of moths mating under different annual grazing intensities expressed as dry stock equivalent (DSE) per hectare.

Grazing intensity	DSE ha^{-1}	Adjustment to intercept (% moths mating at t=0)
Low	1.5	-25.23
Medium	2.3	0
High	5.6	+9.87

Predator control

Expert elicitation suggested that adult GSM are impacted by two types of predators that could be the subject of control measures: birds and cats. Bird predation is enhanced by the presence of perches and thus the control measure is to remove perches that are deemed unnecessary for management or ecological reasons (Table 7.3). We examined aerial images of the small and large conservation areas to determine the approximate number of perches for each reserve size. These were then classified as either removable or irremovable and the area over which enhanced predation might operate because of these two types of perches was calculated. We assumed that enhanced bird predation operates within 50 m of a perch and the average survival under enhanced bird predation was estimated as 58.1% of background survival. We were then able to adjust the proportion of adults surviving to successful mating under different management scenarios (Table 7.4).

Table 7.3: Types of perches and the area affected by enhanced predation by reserve size.

Type	Examples	Area affected small reserve (%)	Area affected large reserve (%)
Removable	Boxthorn, paddock trees, wind rows, windmills	23.6	23.6
Immovable	Internal and external fences, Power pylons, desirable remnant and planted trees	61.3	17.7

Cat control was considered in small reserves only. The model does not assume a particular control method, only that cats are prevented from accessing reserves. The expert elicited adjustment to the background rate of survival to successful mating was 77.4%.

Table 7.4: Management scenarios for predator control.

Predator	Control measure	Base scenario	Management scenario
Birds	Remove perches	Enhanced predation operates within 50m of all perches	Enhanced predation operates only within 50m of immovable perches
Cats (small reserves only)	Remove domestic cat access	Domestic cat predation due to unrestricted access	Domestic cats have no access to site
Bird plus cat (small reserves only)	Remove perches Remove domestic cat access	Enhanced predation operates within 50m of all perches And Domestic cat predation due to unrestricted access	Enhanced predation operates only within 50m of irremovable perches And Domestic cats have no access to site

The adjustments to adult mating for each predator control scenario and reserve size are shown in Table 7.5.

Unofficial

Table 7.5: Adjustment to survival of adults to successful mating weighted by areas containing removable and irremovable perches.

Reserve size	Percentage of area affected (no perch removal)	Adjustment to survival (cat control)	Adjustment to survival (no cat control)	Percentage of area affected (perch removal)	Adjustment to survival (cat control)	Adjustment to survival (no cat control)
Small	84.9	0.644	0.498	61.3	0.743	0.575
Large	41.3	NA	0.827	17.7	NA	0.926

7.2.7 Sensitivity analysis

All three sensitivity analysis approaches were employed to examine the effect of changes in the vital rates on the growth rate and the ranking of management actions. Deterministic sensitivity analysis was conducted by systematically varying one parameter at a time to see the effect of this change on the growth rate. Stochastic analysis involves incrementally increasing or decreasing the parameter estimates by 10% to see the effect of this change on the growth rate. The scenario based sensitivity analysis explored the options of nonlinear relationships (quadratic and exponential) between fecundity and biomass control, and how these scenarios effect the ranking of the management actions.

7.3 Results

7.3.1 Management actions

The ranking of management actions varies according to the nature of the management actions and the size of the reserves they are being applied to. Two types of control measures (biomass control and predator control) are applied concurrently in small and big reserves. Expected minimum abundance (EMA) of GSM populations is the response variable that is used to assess the effectiveness of management actions.

In big reserves, high and medium intensity grazing combined with removal of perches are the most beneficial management actions to maximise the EMA (Figure 7.5).
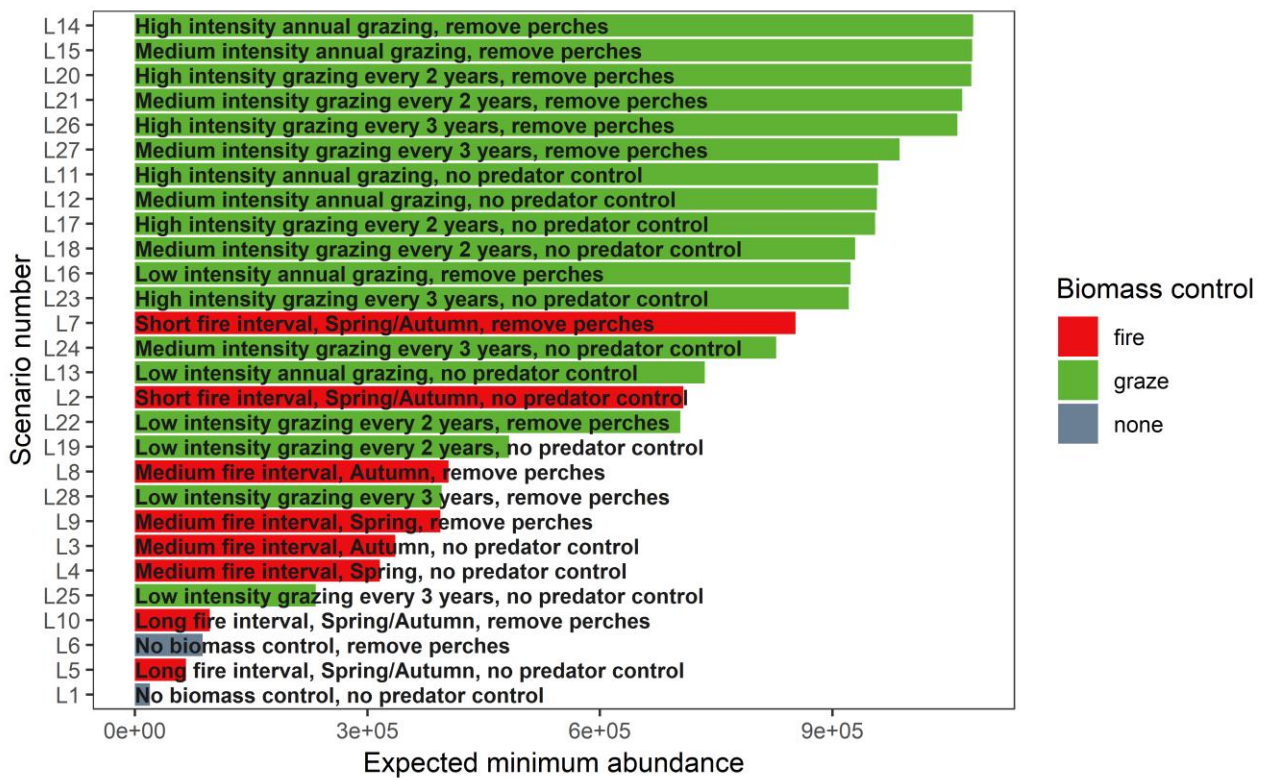
Figure 7.5: Ranking of the benefits of each Golden Sun Moth management scenario for big reserves. Benefits are expressed as the proportional increase in EMA compared to 'do nothing' scenario.

In small reserves, short planned burns in autumn or spring combined with removal of perches and cat control are the best management actions (Figure 7.6) to achieve the highest EMA.
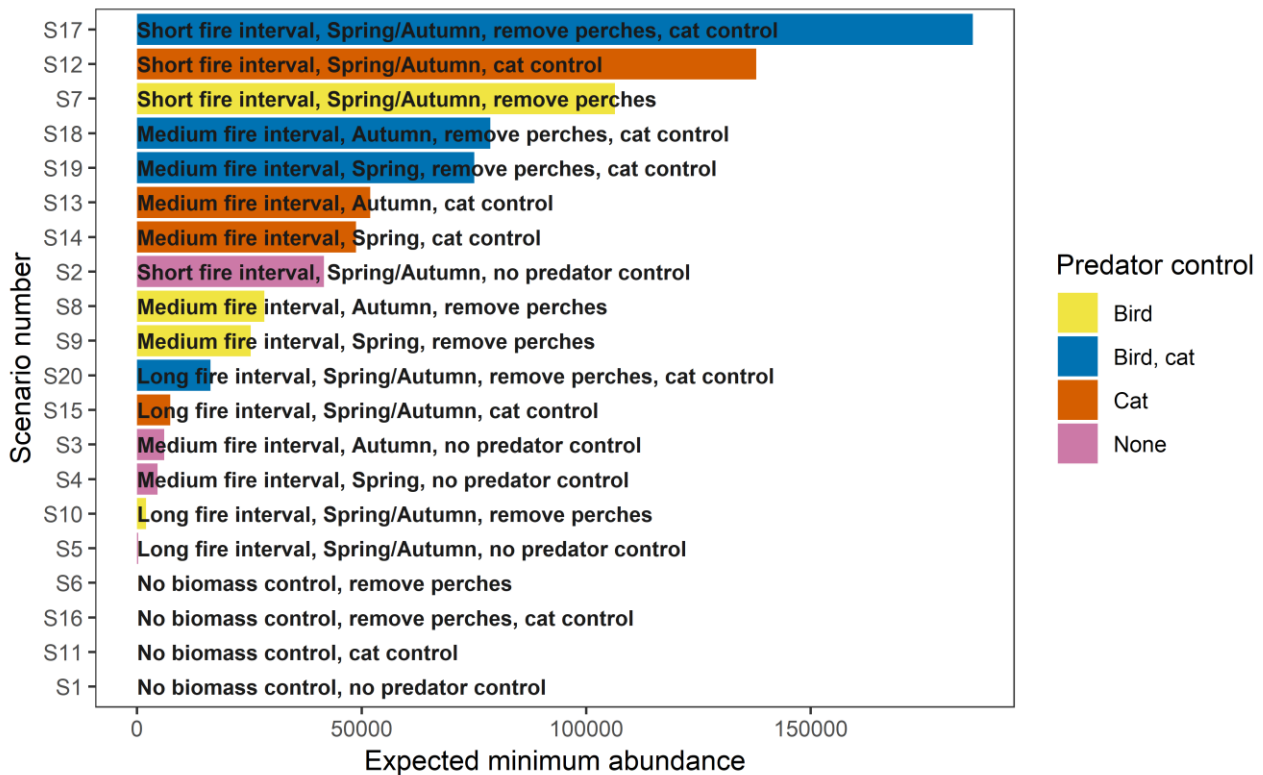
Figure 7.6: Ranking of the benefits of each Golden Sun Moth management scenario for small reserves. EMA expressed in raw value, for baseline 'do nothing' scenario results in zero EMA.

7.3.2   Sensitivity Analysis

Deterministic sensitivity analysis

Deterministic sensitivity analysis suggests that 'adult emergence' and fecundity are the most important population parameter according to both measures of deterministic analysis (i.e. elasticities). These vital rates have the highest proportional contribution to the growth rate. (Figure 7.7).
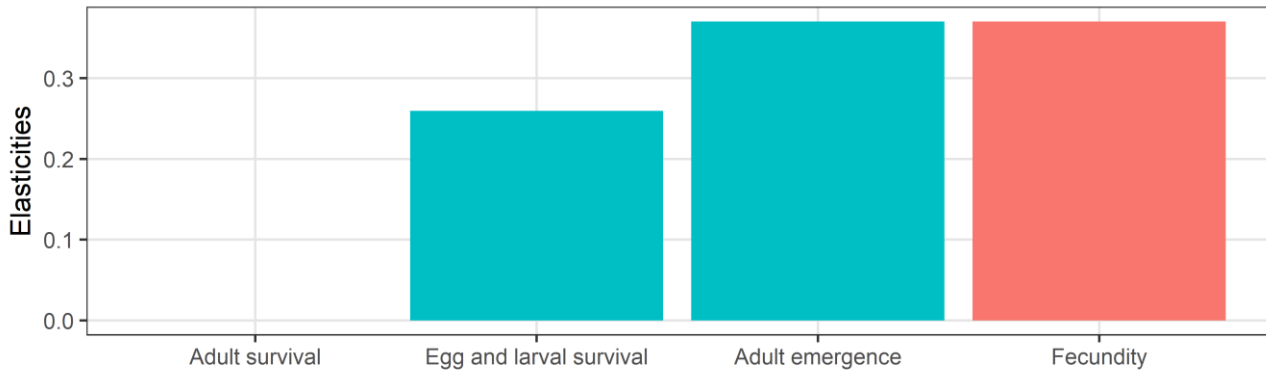
Figure 7.7: Results of deterministic sensitivity analysis. Elasticities of each life stage matrix parameter ranked in ascending order.

Stochastic sensitivity analysis

A model is sensitive to an input parameter if a certain percentage change in the parameter results in a larger percentage change in the output of interest (i.e. EMA). Stochastic sensitivity analysis revealed that the model is particularly sensitive to positive and negative changes to survival and fecundity and moderately sensitive to an increase in carrying capacity and an increase in the standard deviation of survival.

We tested the impact of ± 10% changes in survival, fecundity, initial abundance, carrying capacity and the standard deviations of survival and fecundity. Comparisons were based on the EMA for the "do nothing" scenario at a large reserve (Figure 7.8)
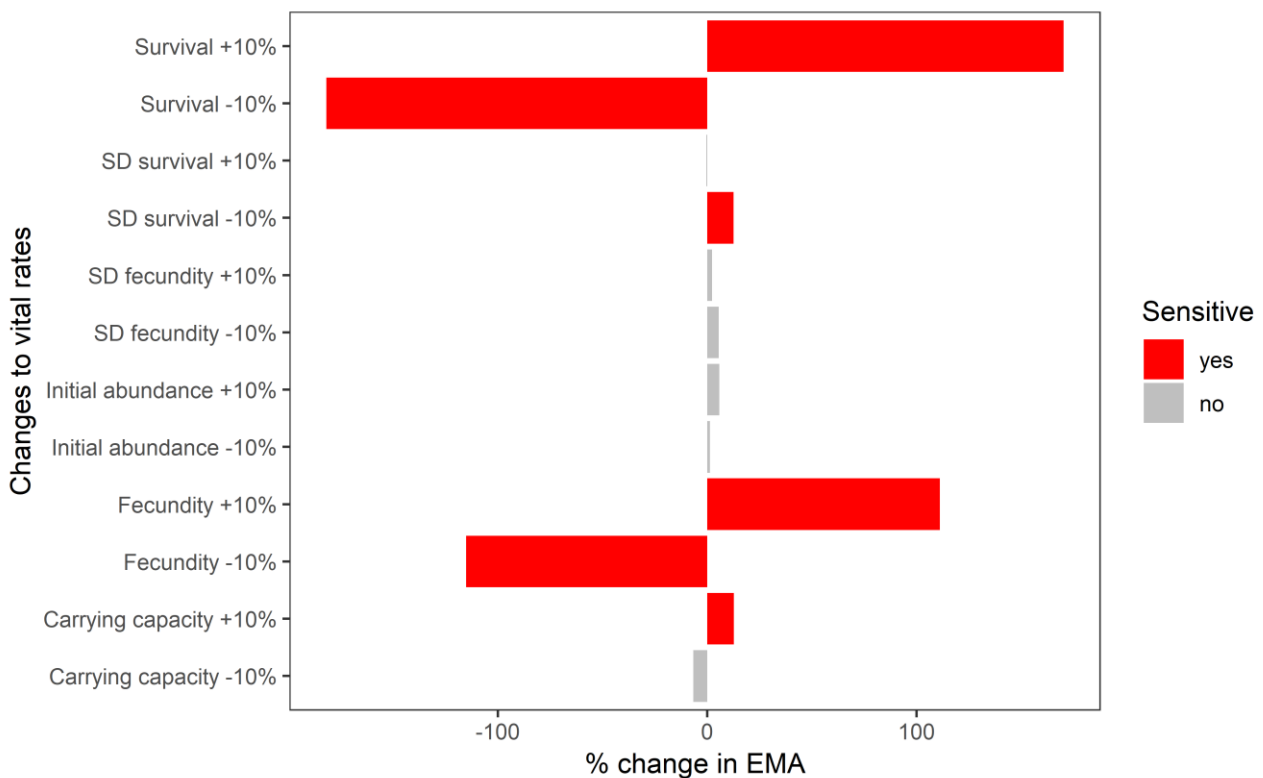
Figure 7.8: Results of stochastic sensitivity analysis. Changes in EMA of greater than 10% (greater than the input change) are coloured red.

The outcomes of stochastic sensitivity analysis suggested that the model is highly sensitive to changes in fecundity. We tested the impact of nonlinear relationships between biomass control measures (medium grazing intensity and fire) and fecundity in large reserves to explore potential alternative outcomes of the model. Fecundity is modelled as proportion of moths mating. Two nonlinear relationships considered in scenario-based sensitivity analysis are:

A. The proportion of moths mating declines rapidly with biomass accumulation at the beginning and then asymptotes (represented by an exponential decay function).

$y = ic^x$

B. The proportion of moths mating declines slowly at first and then becoming more rapid as biomass accumulates (represented by a negative quadratic function).

$y = -cx^2 + i$

where i = intercept of linear relationship and 0< c < 1, a constant specific to the biomass control, x (≥ 0) is the time since biomass control in years.

We also examined the impact of fire removing more biomass, equal to that of high intensity grazing, the parameters for these relationships are shown in Table 7.6.

Table 7.6: Impact of different relationships between time since biomass control and the proportion of moths mating.

Control	Relationship	*i*	*c*
Medium intensity grazing	Exponential decline	0.8838	0.837
Fire	Exponential decline	0.9395	0.73
Medium intensity grazing	Negative quadratic	0.8838	0.00325
Fire	Negative quadratic	0.9395	0.0135

The results suggest that the model is not sensitive to assumptions about the shape of the fecundity-biomass relationship (Table 7.7).

Table 7.7: Results of scenario sensitivity analysis. Comparison of rank order of two types of functional form models compared to base model ranks

Management scenario	Base models	Exponential decline function	Negative quadratic function
	EMA (rank)	EMA (rank)	EMA (rank)
Short fire interval, Spring/Autumn, remove perches	840128 (4)	523848 (4)	944139 (4)
Medium fire interval, Autumn, remove perches	399913 (5)	177854 (5)	556927 (5)
Long fire interval, Spring/Autumn, remove perches	95325 (6)	19243 (6)	155649 (6)
Medium intensity annual grazing, remove perches	1080182 (1)	1081408 (1)	1080084 (1)

Unofficial

Medium intensity grazing every 2 years, remove perches	1068671	863102	1078535
	(2)	(2)	(2)
Medium intensity grazing every 3 years, remove perches	980665	537341	1071548
	(3)	(3)	(3)
Correlation coefficient		1	1

7.4   Discussion

We modelled two population sizes for GSM, representing populations in large management units within the Western Grasslands Reserve and smaller isolated conservation reserves. Within the Western Grassland Reserve the best strategy was high intensity annual pulse grazing of no longer than 4 weeks with perch removal. However, similar benefits were observed for medium intensity annual grazing and high or medium intensity grazing every two years, all with perch removal. The favouring of grazing over fire is largely because it can be done at higher frequencies, thus achieving more frequent biomass removal, benefiting moth fecundity through more mating opportunities. Grazing may also advantage C3 grasses (the preferred food grasses) over C4 grasses. Medium intensity annual grazing occurring over two to three months per year is assumed to be broadly consistent with the most recent management regime on properties where the species persists in high numbers. It should be noted that frequent fire (every 1-3 years) is also beneficial. Removal of perches, whilst providing some benefit, is a lower priority than maintaining low biomass, the cost of removal should be factored into any decision, some perch removal such as boxthorn control will likely be undertaken as part of regular weed control activities. For smaller populations where grazing is not feasible, frequent fire is the best action, with perch removal and cat control undertaken. Cat control, which may not currently be feasible gives a greater benefit than perch removal. However, perch removal, is relatively more beneficial in small reserves compared to large reserves, due to greater perch density.

Sensitivity analysis revealed that the model was most sensitive to adult emergence and fecundity. We investigated the fecundity relationship in scenario sensitivity analysis by assuming different shaped relationships between time since biomass reduction and moth mating percentage. This showed that the ranking of biomass management scenarios wasn't sensitive to the shape of this relationship. This is likely because if biomass control is frequent enough the difference between these relationships is small.

Our models make several assumptions that are either based on expert judgements or a limited amount of data. We tested two predator control scenarios, bird control (via the removal of perches) and cat control. The effectiveness of these actions was entirely based on expert judgement and therefore represent a research gaps. Very little is known about pre-adult life stages in this species, as such we cannot model the impacts of management actions on these life stages, and we were unable to test any scenarios impacting these stages. One of our major assumptions was that the main food plants for larvae are perennial C3 grasses, both native and exotic. Whilst this is based on observations of moths in exotic grasslands (Brown et al. 2012), the extent to which they eat exotic grasses is unclear. Therefore, there are potentially negative impacts of weed control in areas with high levels of exotic C3 grasses such as Nassella trichotoma. Furthermore, the use of herbicides may have direct physiological effects on insects (Gill et al. 2018).

Our models suggest that high intensity annual grazing is the best management scenario on large reserves (where grazing is appropriate), this a deviation from recent historical regimes, which are assumed to be closer to medium intensity. High intensity grazing is likely to be detrimental to other ecological values of interest to the MSA, particularly the structure and composition of natural temperate grassland and the persistence of EPBC listed plants and striped legless lizard. Furthermore, it is unclear what the long-term effects of frequent high-intensity grazing would have on GSM. Potential negative impacts include soil compaction and removal of food plants. Currently the model does not take these into account. Given that GSM has persisted in the Western Grassland Reserve under grazing, the continuation of moderate intensity grazing is most conservative until we acquire information that suggests otherwise. In small reserves, where grazing is not appropriate, frequent fire was the best biomass control action. As these models were largely expert driven our understanding of how GSM responds to different management actions would be improved by collecting basic life history data, particularly related to the behaviour of larvae underground.

8 Southern Brown Bandicoot (*Isoodon obesulus obesulus*)

8.1 Background

The Southern Brown Bandicoot, *Isoodon obesulus obesulus*, is a medium-sized ground-dwelling marsupial found in Victoria and other parts of south-eastern Australia. Adult southern brown bandicoots weigh 400-1850 grams, have long tapered snouts, a compact body and short, pointed tail. They feed predominantly on invertebrates, mainly insects and earthworms, but also small vertebrates (Brown and Main 2010), plants and fungus (Broughton and Dickman 1991).

In Victoria, the Southern Brown Bandicoot (SBB) is primarily found in coastal regions, including the south-east Melbourne region (Brown and Main 2010). Isolated populations are also found in the Dandenong Ranges, the Grampian Ranges and central western Victoria (Menkhorst 1990). Within the MSA area, SBB is found within the south-east of the urban growth boundary and adjacent areas outside the growth boundary. There is a large population in Cranbourne Botanic Gardens. Suitable habitat is located to the east of the growth boundary but is generally fragmented (DSE 2009). SBB is found in a variety of habitats including woodland, heathy open forest, sedgeland, shrubland and heathland, usually areas with plenty of cover to protect them from predators (Menkhorst 1990).

The main threats to SBB in the area include habitat destruction through urban expansion, the effects of fragmentation of the remaining isolated habitat, and predation from foxes. There is debate as to how best to manage the species within the urban growth boundary and surrounds ranging from the development of corridors, baiting foxes, habitat creation and creating conservation reserves and reintroducing bandicoots. Several PVAs have been developed for the species over the last decade to investigate different questions related to the extinction risk and management of SBB (e.g.Possingham and Gepp 1993, Southwell et al. 2008). These PVA models, along with existing literature and expert judgements were reviewed and integrated to develop a spatially explicit PVA model for SBB within the MSA management area. Unlike other species in this report, the model developed was spatially explicit because the management levers investigated were inherently spatial. This model was used to estimate the relative benefits of different alternatives on the persistence of the bandicoots within the MSA management area and surrounds. Multiple alternatives focusing on enhancing connectivity through the development of corridors or creating a connectivity network of habitat stepping-stones, baiting foxes and reintroducing bandicoots to reserves and combinations of these actions were explored.

8.2 Methods

8.2.1 Conceptual models

Two conceptual models were developed for SBB to illustrate the habitat and population dynamics for SBB and how threats and management influence them. The first conceptual model illustrates the life cycle of the species, threats to population processes and where management can intervene (Figure 8.1). The life cycle of SBB can be represented by four ages classes; juveniles, 1 year old, 2 year old and 3 year old. It is thought that bandicoots do not live much older than 4 years. Foxes are the main predator of bandicoots and can reduce the survival of all age classes. Baiting foxes and the construction of predator proof fences are two management options to ameliorate the threat of foxes. Demographic and environmental stochasticity also influence the population dynamics of the species. While this variation cannot be managed directly, it is important to account for it in any management of the species.

A second conceptual model depicts the spatial metapopulation across the management area (Figure 8.2). Habitat patches may be occupied or unoccupied, and individuals can disperse from one population to another if the distance between patches is within an individual's dispersal capabilities. Management actions associated with the spatial view include creating habitat corridors or stepping-stones to facilitate dispersal between occupied and unoccupied patches and to improve gene flow, improving habitat for the species by creating new patches of habitat or enhancing existing habitat. Instead of creating corridors to facilitate dispersal, translocating individuals to areas of unoccupied habitat is another management option to populate unoccupied patches of habitat.

Unofficial
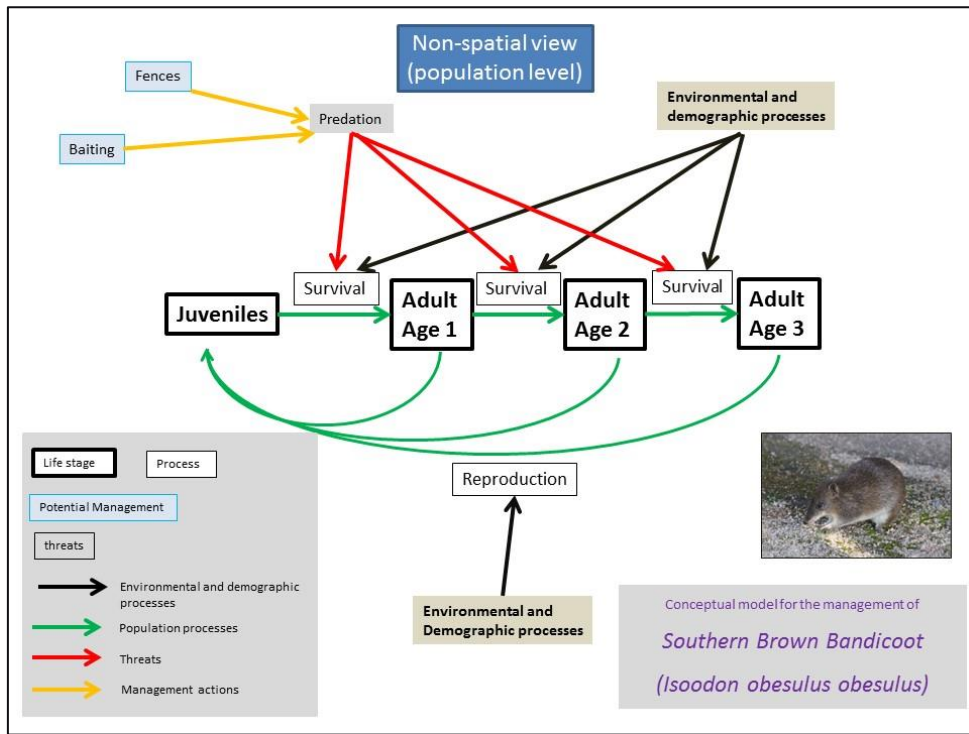
Figure 8.1: Conceptual model of the life cycle of Southern Brown Bandicoot and how threats, management and stochasticity influence key processes.
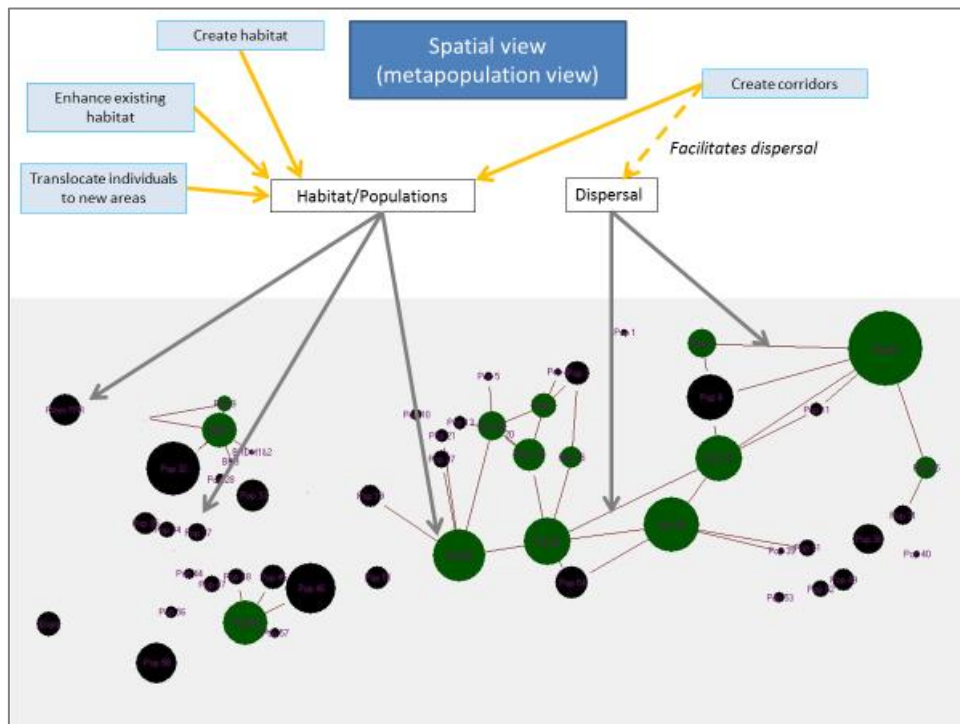
Figure 8.2: Conceptual model of the metapopulation of Southern Brown Bandicoot, highlighting occupied (green circles), unoccupied habitat (black circles), potential pathways of dispersal (lines) and how management can influence them.

8.2.2 Habitat distribution model

The most up to date state-wide habitat distribution model (HDM) for SBB was considered in the analysis. The HDM was used to describe the spatial component of the PVA model. The modelled was clipped to the management extent and habitat thresholds and a mask was applied based on species experts' view of SBB habitat within the management area (Figure 8.3).
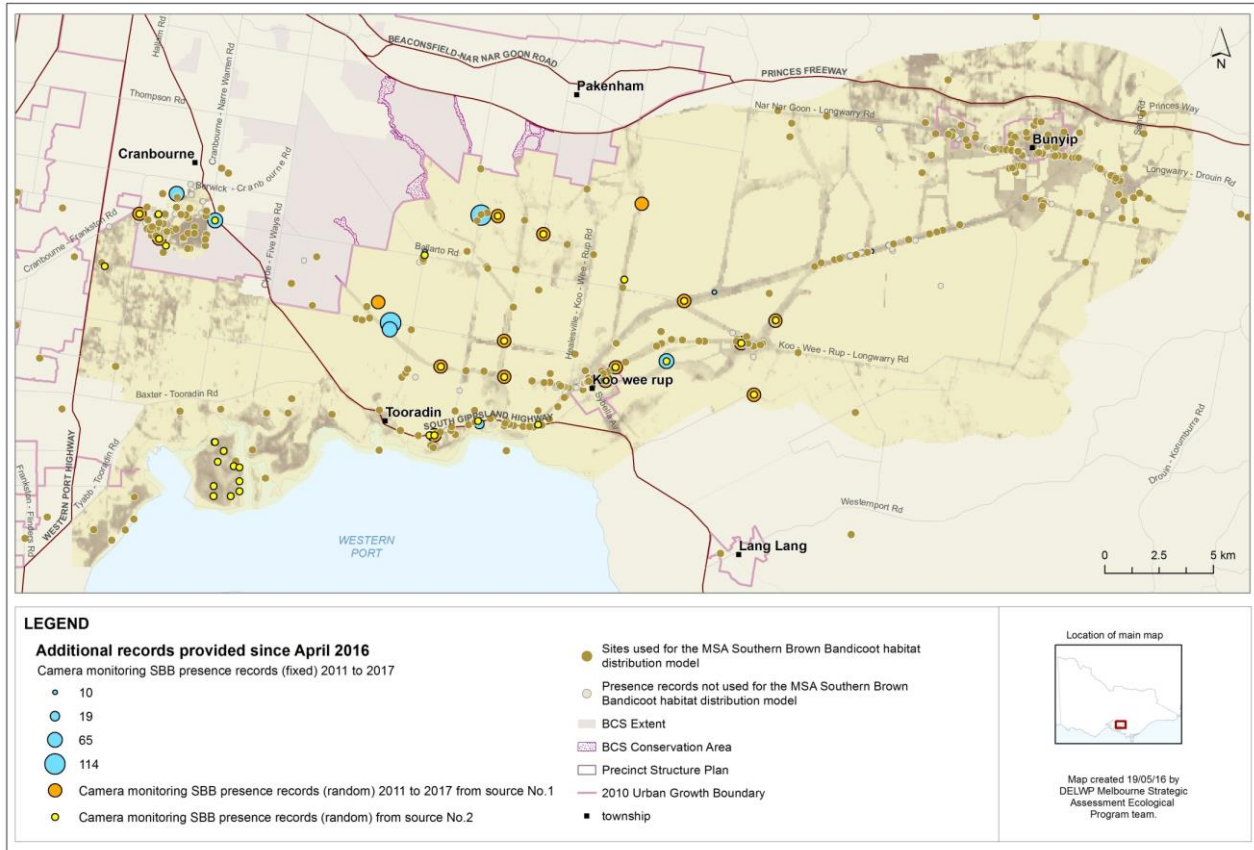
Figure 8.3: Expert derived Habitat Distribution Model for Southern Brown Bandicoot including presence records. Grey areas depict SBB habitat within the management area.

8.2.3 Vital rates

Existing PVA models were reviewed by ecological modellers and species experts to determine their accuracy and relevance for this study. Part of the model structure (i.e. stage matrix) from previous models and existing estimates for fecundity were used for this model, but more recent data on survival rates were available to estimate survival under background predation rates (S. Maclagan, pers. comm.). Estimates for survival given different forms of fox baiting and predator proof fences were not available and were estimated through a structured expert elicitation process with six experts.

Bandicoots are highly fecund producing on average 6.37 offspring per year (Lobert, 1985). Survival rates under fox predation result in an approximately stable growth rate of $\lambda=1.0024$. Survival rates given various types of predator management including, targeted baiting, broad scale baiting and predator proof fences and baiting resulted in growth rates of $\lambda=1.17$, 1.32 and 1.59 respectively (Table 8.1). Bandicoots are thought to only live for four years in the wild, hence Adult 3 (i.e. 4 year old individuals) have a survival rate of zero.

Unofficial

Table 8.1: Survival rates based on existing data and combined expert judgement given different types of management associated with fox baiting

Survival rates	With predation	Targeted fox baiting	Broad scale fox baiting	Fenced and baited
Juveniles	0.19	0.27	0.34	0.51
Adult 1	0.5	0.55	0.61	0.68
Adult 2	0.33	0.35	0.41	0.45
Adult 3	0	0	0	0
Growth rates	1.0024	1.17	1.32	1.59

8.2.4 Stochasticity

The co-efficient of variation (CV), for fecundity and survival rates under fox predation were estimated from data with the CV for fecundity estimated as: 20% and for survival under fox predation as 30% (Lobert 1985), S. Maclagan, pers. comm). For all other survival rates, CVs were estimated from expert elicitation. These were 22% for targeted fox baiting, 23% broad scale fox baiting, 28% for predator proof fences and baiting.

8.2.5 Dispersal

Bandicoots are able to disperse long distances (~6km) along habitat corridors within suitable habitat but less likely to disperse through areas of unsuitable habitat. Bandicoots are known to move through unsuitable areas but in close proximity to suitable habitat. Dispersal was modelled using a distance function where the dispersal rate declines as distance between suitable patches increases (Figure 8.4). This function governs movement of individuals from one patch of suitable habitat to another. Dispersal rates were estimated from published literature (Li et al 2015), and consultation with species experts. Juveniles are more likely to disperse than adult bandicoots. Of the individuals that disperse in a year, 85% are juveniles and 15% are adults.
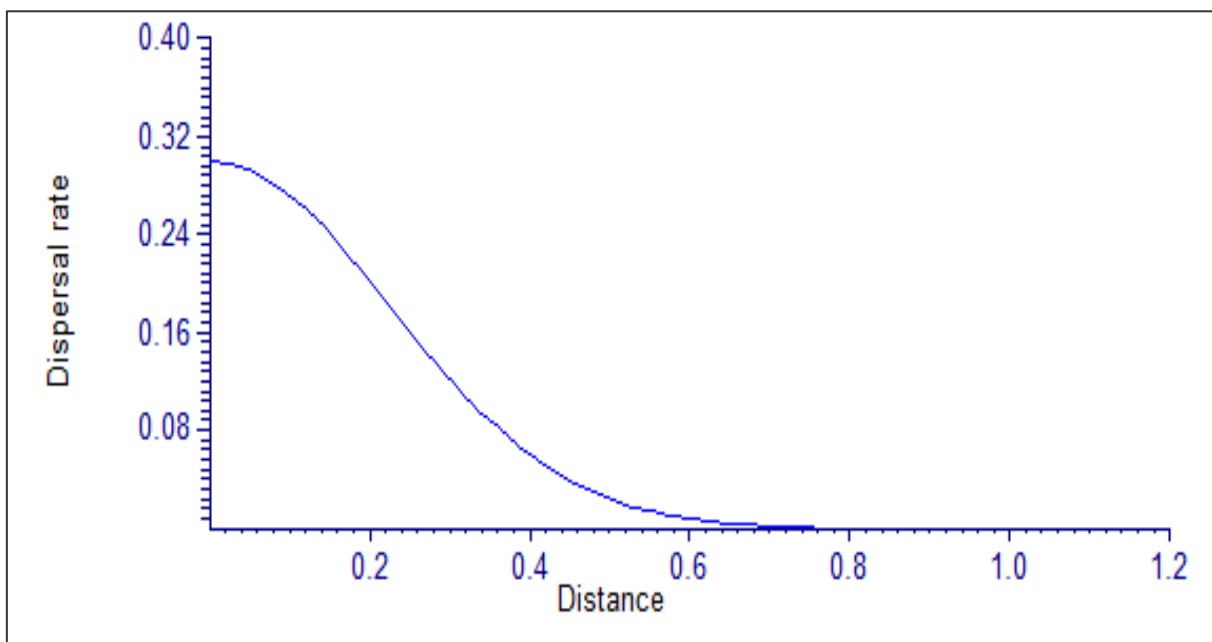

Figure 8.4: Distance dispersal function through unsuitable habitat for Southern brown bandicoots.

8.2.6 Additional model specification

The HDM was translated to a patch map and linked to the population model by specifying several relationships. Firstly, neighbourhood distance defines whether cells of suitable habitat were part of the same patch or not. Neighbourhood distance was derived from the home range of bandicoots (1.84ha). Any suitable cells that were separated by a distance less than the neighbourhood distance were regarded to be in the same patch. Carrying capacity of each patch were calculated as the total habitat suitability of the patch multiplied by the density of bandicoots in optimal habitat: 1.5/ha. Only patches that had a presence record were deemed occupied. Initial abundances of occupied patches were calculated based on an expert estimate of the total population size of the management area of approximately 3000 bandicoots. Patches were linked through the dispersal function that governed the rate of individuals dispersing to neighbouring patches.

Four stage matrices were constructed using the fecundity estimates and vital rates from Table 8.1. Each patch was assigned a stage matrix that reflected the current or proposed management for the model scenario (Table 8.2). For instance, the patch that represents the Cranbourne Botanic Gardens had the fenced and baited stage matrix associated with it (Table 8.1). Density dependence was incorporated as a ceiling model. In this type of model populations can grow exponentially according to the implied growth rate until they reach a ceiling, and then remain at that level. All model scenarios run over 50 years for 1000 iterations. Model scenarios were compared using expected minimum abundance (EMA), the average of the minimum values for each trajectory of a stochastic population model. It is useful summary statistic for comparing management scenarios for species at risk of extinction (McCarthy and Thompson, 2006). Benefits from management actions were compared to a do nothing scenario (i.e. Action 0).

8.2.7 Management scenarios

Consultation with species experts and stakeholders developed a list of 20 alternative management actions (Table 8.2). The actions were divided into broad themes associated with the development of corridors within the urban growth boundary (UGB), corridors that extend outside the UGB, fox baiting, habitat connectivity network, enhancement of existing corridors and reintroductions into existing or previous reserves.

It is assumed all corridors are created according to agreed specifications. Corridors are modelled as separate linear habitat patches that bandicoots can easily disperse to. We assume that corridors also act as habitat such that bandicoots not only move through but can reproduce and forage for food. All areas of new habitat including corridors assume habitat suitability of best quality patches identified in the HDM (i.e. 0.8). Fox baiting alternatives assume two levels of compliance, 25% and 5% (actions 7-8) and do not distinguish between public and private land. No fox baiting occurs in urban areas. Targeted baiting only occurs in currently occupied patches.

Alternative actions focusing on the habitat connectivity network (action 10 and 10a) includes land acquisition or incentive programs that include habitat creation with and without full compliance targeted baiting. Habitat enhancement of existing corridors also includes targeted fox baiting but only at 25% compliance (action 11). Reintroductions assume patches are fenced and baited and habitat is created and managed, and individuals are moved from Royal Botanic Gardens Cranbourne (RBGC), Koo Wee Rup (KWR), or outside the management area (actions 12-17). Alternative management actions do not include feasibility and costs.

Table 8.2: Alternative management actions

Action	Description	Details of action
Action 0	Do nothing	Includes drainage line managed by Melbourne Water. RBGC is fenced and baited in all scenarios. All other habitat patches assume no baiting.
Action 1	Corridors within the UGB	Creation of corridors within UGB as specified in proposed actions draft (option1). No fox baiting or predator proof fences in corridors within the UGB
Action 2	Corridors within the UGB	Creation of corridors within UGB as specified in proposed actions draft (option2). No fox baiting or predator proof fences in corridors within the UGB

Unofficial

Action 3	Corridors within the UGB	Creation of corridors within UGB as specified in proposed actions draft (option3). No fox baiting or predator proof fences in corridors within the UGB
Action 4	Corridors within the UGB	Creation of corridors within UGB as specified in proposed actions draft (option 4). No fox baiting or predator proof fences in corridors within the UGB
Action 5	Corridors outside UGB	Extended corridors outside UGB to Pines, Tyabb and railway reserve. Extended corridors have width of 30m. Corridors inside UGB are as per Action 1. Assumes Pines Reserve is fenced and baited. No baiting within any corridors
Action 6	Corridors outside UGB	Extended corridors outside UGB to Pines, Tyabb and railway reserve. Extended corridors have width of 50m. Corridors inside UGB are as per Action 1. Assumes Pines Reserve is fenced and baited. No baiting within any corridors.
Action 7	Broad scale fox baiting (25% compliance)	Broad scale baiting assumes baiting in SBB habitat and non SBB habitat areas within baiting extent. Broad scale fox baiting is in all patches that overlap baiting extent (see mask). If any part of a patch overlaps any part of the baiting extent, its assumed the whole patch is baited. It is assumed that compliance of broad scale baiting is 25%. This does not distinguish between public or private land.
Action 7a	Broad scale fox baiting (5% compliance)	Broad scale baiting assumes baiting in SBB habitat and non SBB habitat areas within baiting extent. Broad scale fox baiting is in all patches that overlap baiting extent (see mask). If any part of a patch overlaps any part of the baiting extent, its assumed the whole patch is baited. Assumes compliance of broad scale baiting is 5%. This does not distinguish between public or private land.
Action 8	Targeted fox baiting (25% compliance)	Targeted baiting only occurs in occupied patches in areas that overlap with the baiting extent. If any part of an occupied patch overlaps any part of the baiting extent, its assumed the whole patch is baited. It is assumed that compliance of broad scale baiting is 25%. This does not distinguish between public or private land.
Action 8a	Targeted fox baiting (5% compliance)	Targeted baiting only occurs in occupied patches in areas that overlap with the baiting extent. If any part of an occupied patch overlaps any part of the baiting extent, its assumed the whole patch is baited. It is assumed that compliance of broad scale baiting is 5%. This does not distinguish between public or private land.
Action 10	Land acquisition and creation of SBB habitat connectivity network	Acquire land parcels and creation of habitat connectivity network. Parcels of land are within dispersal distance of SBB and link to known occupied patches. Habitat is created to a habitat suitability of 0.8. No fox baiting or predator proof fences
Action 10a	Land acquisition and creation of SBB habitat connectivity network with targeted fox baiting.	Acquire land parcels and creation of habitat connectivity network. Parcels of land are within dispersal distance of SBB and link to known occupied patches. Assumes full compliance of targeted fox baiting in acquired areas. Habitat is created to a habitat suitability of 0.8. No predator proof fences
Action 11	Habitat enhancement	Enhancement of habitat along utility corridors. Habitat is improved to habitat suitability of 0.8. Assumes targeted fox baiting in enhanced

	within existing utility corridors	patches at 25% compliance.
Action 12	Reintroduction From RBGC to Pines reserve	Assumes fence and baiting at Pines. Capture and relocate 20 SBB from RBGC to Pines in year 1 (10 male, 10 female). Capture and relocate 10 SBB from KWR to Pines in year 5 (5 male, 5 female). Capture and relocate 10 SBB from KWR to Pines in year 10 (5 male, 5 female)
Action 13	Reintroductions from outside to Pines reserve	Assumes fence and baiting at Pines. Capture and relocate 20 SBB from outside to Pines in year 1 (10 male, 10 female).Capture and relocate 10 SBB from outside to Pines in year 5 (5 male, 5 female). Capture and relocate 10 SBB from outside to Pines in year 10 (5 male, 5 female)
Action 14	Reintroduction from RBGC and KWR to Briars	Assumes fence and baiting at Briars. Capture and relocate 20 SBB from RBGC to Pines in year 1 (10 male, 10 female). Capture and relocate 10 SBB from KWR to Pines in year 5 (5 male, 5 female). Capture and relocate 10 SBB from KWR to Pines in year 10 (5 male, 5 female)
Action 15	Reintroduction from outside management area to Briars	Assumes fence and baiting at Briars. Capture and relocate 20 SBB from outside to Briars in year 1 (10 male, 10 female). Capture and relocate 10 SBB from outside to Briars in year 5 (5 male, 5 female). Capture and relocate 10 SBB from outside to Briars in year 10 (5 male, 5 female)
Action 16	Reintroduction from RBGC and KWR to Pines and Briars	Assumes fence and baiting at Pines and briars. Capture and relocate 20 SBB from RBGC and KWR to Pines and Briars in year 1 (10 male, 10 female each). Capture and relocate 10 SBB from RBGC and KWR to Pines and Briars in year 5 (5 male, 5 female each). Capture and relocate 10 SBB from RBGC and KWR to Pines and briars in year 10 (5 male, 5 female each)
Action 17	Reintroduction from outside management area to Pines reserve and Briars	Assumes fence and baiting at Pines and briars. Capture and relocate 20 SBB from outside to Pines and Briars in year 1 (10 male, 10 female each). Capture and relocate 10 SBB from outside to Pines and Briars in year 5 (5 male, 5 female each). Capture and relocate 10 SBB from outside to Pines and Briars in year 10 (5 male, 5 female each)

8.2.8   Sensitivity analysis

Three types of sensitivity analysis were investigated: deterministic, stochastic and scenario sensitivity. The deterministic sensitivity analysis focused on the sensitivities and elasticities of the stage matrix. Elasticities reflect the proportional contribution of each vital rate to the overall growth rate. Sensitivities are indices of the influence of small changes in vital rates on the growth rate. The stochastic sensitivity analysis involved adjusting key model parameters by ±10% to investigate if the stochastic model was sensitive to any these parameters. If the EMA results changed by more than ±10% this indicates the model is sensitive to that parameter. Parameters tested include all vital rates (i.e. fecundity and survival), standard deviations of vital rates, initial abundance, dispersal, and carrying capacity.

The scenario sensitivity analysis evaluated how robust the rank order of alternative actions is to key model assumptions. Firstly, the most sensitive parameter from the stochastic sensitivity was investigated to see if it influences the rank order of management actions. Secondly, the dispersal capabilities of SBB are largely unknown and the distance dispersal function was derived based on limited evidence and expert judgement. The influence of dispersal on the rank order of actions was analysed using two extreme dispersal scenarios, no dispersal between habitat patches, and a high rate of dispersal as depicted in Figure 8.5. The scenario sensitivity analysis was done for a subset of eight management actions across all action types.
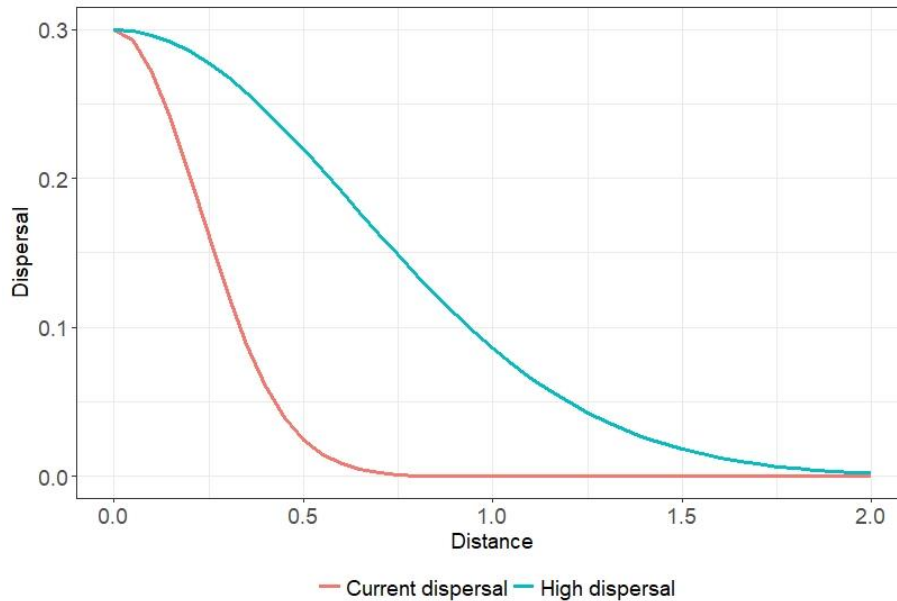
Figure 8.5: Two scenarios for dispersal explored with the model, current dispersal (red) and a high rate of dispersal (blue).

8.3 Results

8.3.1 Patch analysis

The patch analysis resulted in 80 separate patches that form the metapopulation for the Southern brown bandicoot within the management area (Figure 8.6). These include all proposed corridors leading out from the RBGC and patches included in the habitat connectivity network. Only 14 patches were initially occupied by bandicoots. These patches represent the known populations of SBB in the area including the RBG, Koo Wee Rup swamp and Bunyip to the east of the management area to name a few. Unoccupied patches can become occupied if bandicoots can disperse to them within the 50-year time frame.
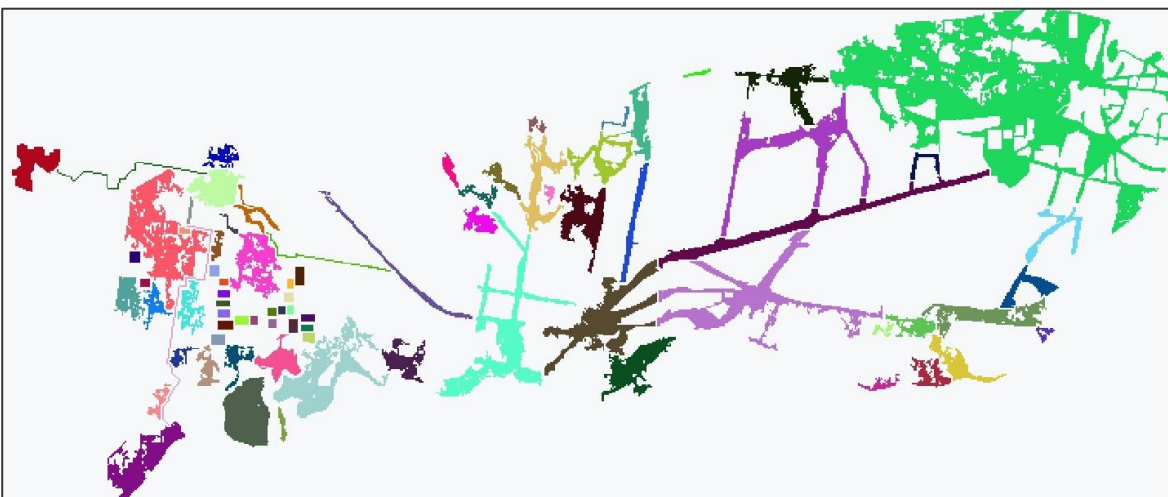
Figure 8.6: Patch map for Southern brown bandicoot in management area including all corridors and habitat connectivity network. Each colour represents a separate occupied or unoccupied patch.

8.3.2   Management actions

Results of relative benefits of alternative management actions revealed two actions having the greatest benefits to SBB persistence (Figure 8.7). Broad scale fox baiting (i.e. action 7) had the highest benefit to SBB persistence. This action requires 25% compliance and continuous baiting over large areas. If compliance is low (i.e. 5%) then the benefits drop substantially (actions 7a and 8a). Creation of a habitat connectivity network or enhancement of existing corridors also provided large benefits but only if coupled with targeted fox baiting (action 10a and 11). Without fox baiting as part of the connectivity network, the benefits reduce to more than half (Action 10). Reintroductions are also beneficial as along as areas are managed, fenced and baited and habitat is suitable for bandicoots. The actions providing the largest benefits all have some form of fox control, indicating that predator control is necessary for managing SBB in the region. Corridors within the UGB provided minimal benefit to bandicoots (actions 1-4). Extending corridors outside the UGB brings higher benefits to SBB especially if corridors are >50m in width (actions 5-6) and the target patch is predator free.
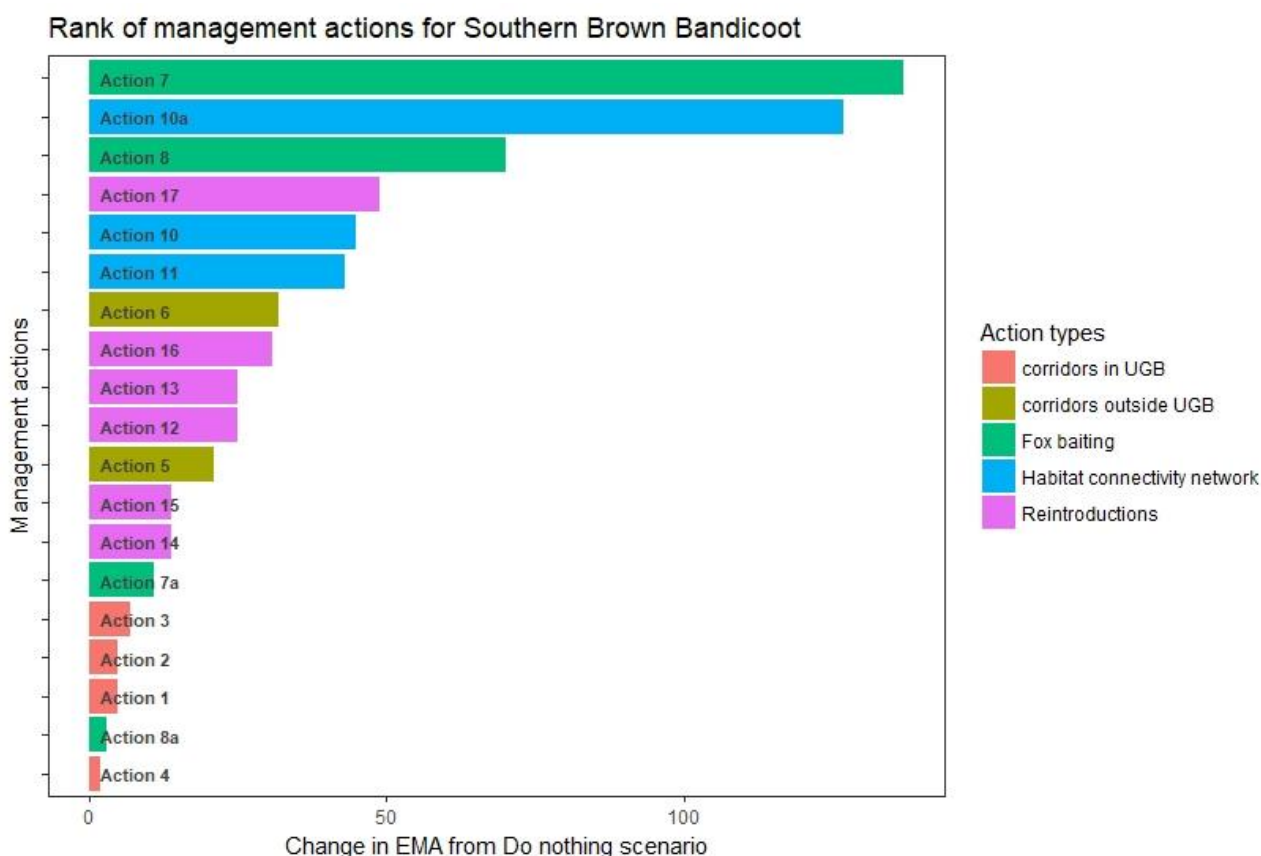

Figure 8.7: Ranks of management actions based on the relative benefit of the action compared to a do nothing scenario. Benefit is measured as the change in expected minimum abundance (EMA).

8.3.3   Sensitivity analysis

Deterministic sensitivity analysis of the stage matrix revealed the growth rate is sensitive to estimates of juvenile survival (Figure 8.8). The sensitivity of this parameter is overwhelming compared to other vital rates. Juvenile survival also contributes to the growth rate the most, highlighting the importance of having accurate estimates for this parameter. The growth rate is also sensitive to the survival of young adults but to a lesser extent than juvenile survival. Fecundity of young adults contributes to the growth rate, but the growth rate is not sensitive to it.

The stochastic sensitivity analysis confirmed the model is most sensitive to survival rates, but also revealed the model sensitivity to fecundity estimates (Figure 8.9). Fortunately, there is good evidence to support the estimates of fecundity of SBB (Lobert 1985). The model was insensitive to changes in all other model parameters tested.

Unofficial

The scenario sensitivity revealed that the ranking of management actions was robust to different assumptions regarding dispersal (Table 8.3). Rank correlations for models with no dispersal and high dispersal were 0.98 and 0.9 respectively with only minor changes in the rank order of actions given different dispersal assumptions. Ranks of management actions were not robust to assumptions regarding survival rates. Rank correlations were 0.62 when survival rates were altered by ± 10%. The poor correlation was driven by the change in rank order of Action 10a: the creation of a habitat connectivity network. This action was originally ranked 2 having a high benefit to bandicoots. When survival rates were altered, this action ranked 7, becoming one the least preferred actions.
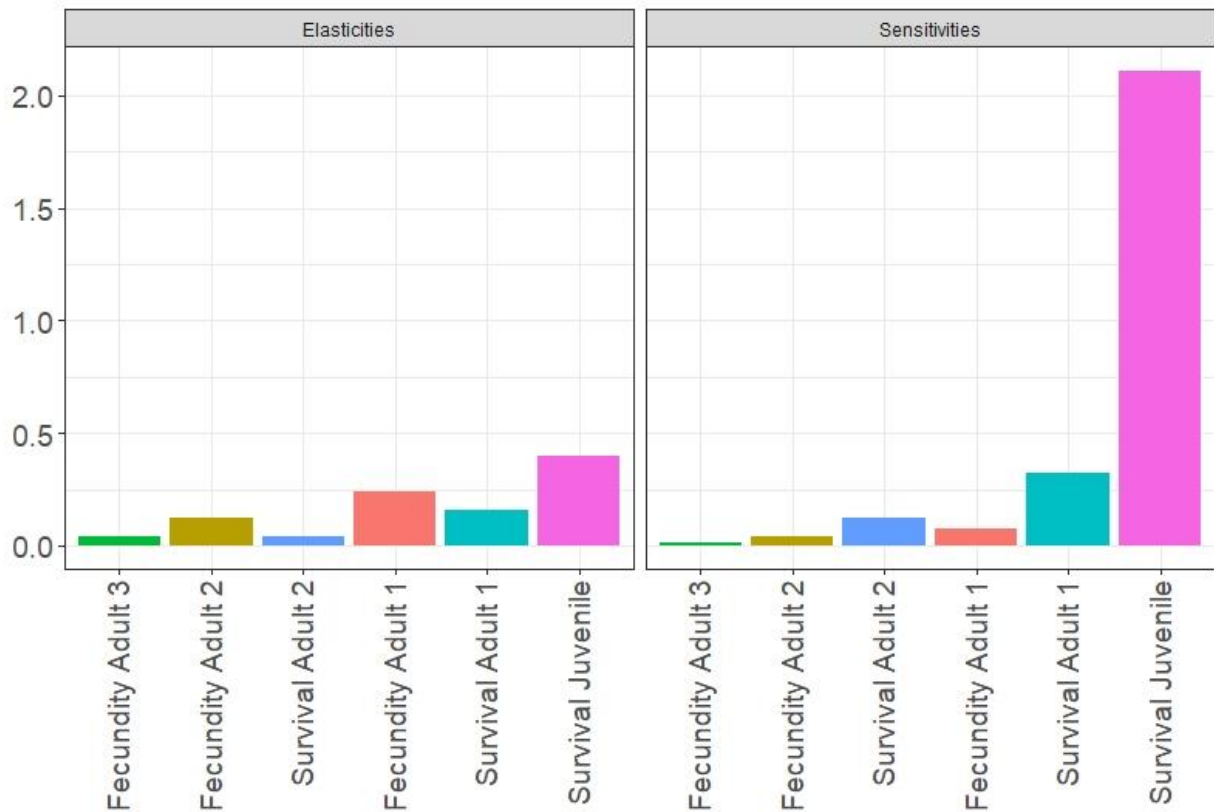
Figure 8.8: Deterministic sensitivity analysis including elasticities and sensitivities of all vital rates
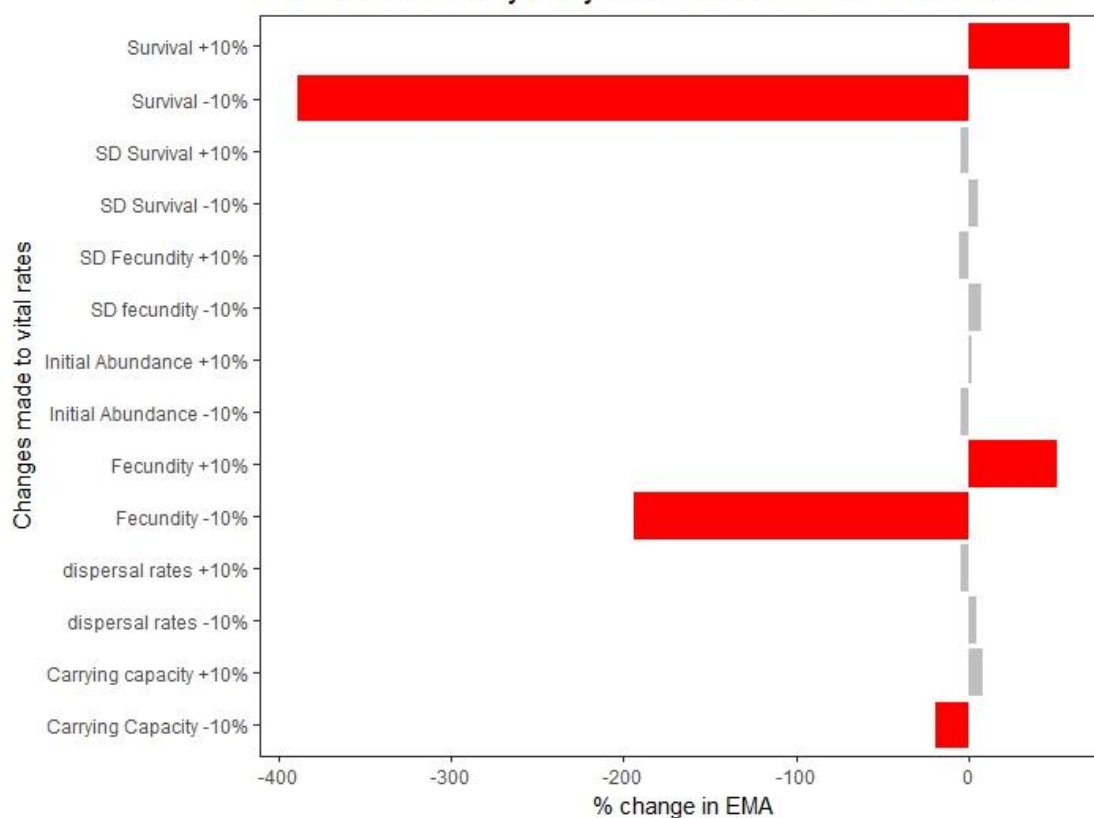
Figure 8.9: Stochastic sensitivity analysis for key model parameters measured as the % change in expected minimum abundance from the base model. Red bars indicate sensitive parameters while grey bars indicate parameters the model is insensitive to.

Table 8.3: Scenario sensitivity analysis indicating rank order and rank correlation between base model and different assumptions about survival and dispersal

	Base model	+10% survival	-10% survival	No dispersal	High dispersal
Action 7	1	1	1	1	2
Action 10a	2	7	7	2	1
Action 8	3	3	4	3	3
Action 17	4	2	2	4	6
Action 10	5	4	6	6	4
Action 11	6	5	5	5	5
Action 6	7	6	6	7	7
Action 3	8	8	8	8	8
Rank correlation with Base		**0.62**	**0.62**	**0.98**	**0.90**

Unofficial

8.4 Discussion

The development of a spatially explicit population viability analysis model for the Southern Brown Bandicoot has improved our understanding of the effectiveness of alternative management options for the species within the MSA management area. Twenty detailed management alternatives were devised by experts and stakeholders and implemented into the model to explore differing opinions about management effectiveness. All actions explored provided benefits to SBB persistence, but to different degrees. Continuous broad scale fox baiting with 25% compliance was the top ranked action for the species. The creation of a habitat connectivity network was also beneficial if coupled with predator control, while corridors within the UGB provided the least benefits to bandicoots. Reintroductions can be beneficial if reserves are fenced and managed appropriately. Results revealed the top ranked actions all include some form of predator control either though high compliance fox baiting or predator proof fences highlighting the importance of predator control to sustain populations of SBB in the region.

The modelling focused on benefits of alternative actions assuming actions are implemented to particular standards. This included continuous fox baiting, habitat creation that results in 0.8 relative habitat suitability and all reserves are fenced and managed appropriately. This study ignores feasibility and cost of actions. In practice some actions may be less effective than assumed here due to implementation issues. For instance, scenarios that include predator control assume continuous baiting across the entire 50-year time horizon. Once off baiting may have initial benefits but SBB survival rates are likely to decline once baiting stops and fox densities increase. Thus if there is no commitment for ongoing predator control then the effectiveness of predator control will not be realized. In this case another action may be more beneficial. Similarly, all scenarios that involve habitat creation assume the highest quality habitat for SBB which translates to a relative habitat suitability index of 0.8. If habitat cannot be created to this standard, then expected benefits will not be realised. When devising management plans for SBB it will be important to consider not only the results presented here but also the feasibility and costs of the actions. This may result in a very different rank order of management actions. While some initial work has been done to estimate feasibility and costs of the actions explored in this study, further exploration of the cost effectiveness and feasibility is needed.

Sensitivity analyses revealed the model is most sensitive to survival rates, particularly juvenile survival. This was evident in all three sensitivity analyses explored. Changes in survival rates also changed the rank order of management actions, particularly the actions associated with creating a habitat connectivity network. This action changed from most preferred to least preferred when survival rates were altered by 10%. This highlights the importance of attaining more accurate estimates of survival. Otherwise it will be preferable to choose actions that are less sensitive to changes in survival rates. Attaining more accurate estimates of survival, particularly given different levels of predator control will be important for targeting the most appropriate management actions. Unfortunately estimating juvenile survival is particularly difficult as juveniles tend to disperse and are harder to trap. Devising better survey methods for estimating juvenile survival should be considered for future research.

Stochastic sensitivity also revealed the model was sensitive to fecundity. Estimates for fecundity were based on several studies so we are confident the estimates used in the model are accurate. Prior to modelling, experts were concerned about uncertain estimates for dispersal, thus we tested the assumption with two extreme alternatives, no dispersal and very high dispersal. Fortunately, the rank order of management actions was robust to our dispersal assumptions, suggesting the uncertainty in dispersal will not influence the choice of management.

The highest benefits for SBB given the management alternatives investigated in this study were achieved through some form of predator control. For the ongoing persistence of SBB within the management area some form of predator control should be considered as part of a management plan. If fox baiting is deemed infeasible due the proximity to urban areas, alternative methods for controlling foxes and other predators should be investigated. This is an area of future research. The spatially explicit population viability model developed here uses the most up to date information on Southern Brown Bandicoot. It is specific to the MSA management area and if further management alternatives are devised they can be explored through this model.

9 Striped Legless Lizard (*Delma impar*)

9.1 Background

The Striped legless lizard (*Delma impar*) is a legless or flap-footed lizard of the family Pygopodidae. It superficially resembles a snake but can be distinguished from co-occurring snake species by the presence of external ear openings, an unforked tongue and the presence of vestigial hindlimbs. Striped legless lizards (SLL) body lengths can reach a maximum of approximately 300 mm. The life history of SLL is largely unknown. Estimates of lifespan are about 10 years (based mainly on observations in captivity), but individuals may live significantly longer (ARAZPA 1996). The clutch size is fixed, with each female laying two eggs and it is likely that most females can breed every year. SLL mostly spend the colder months in soil cracks, under or beside rocks and in tussock bases. In Victoria, SLL is known to be active between August and December with most activity between September and October.

SLL is listed as Vulnerable under the Commonwealth's Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act), Threatened under Victoria's Flora and Fauna Guarantee Act and Endangered in the DELWP advisory list of threatened fauna (DSE 2013). SLL is patchily distributed in native grasslands, grassy woodlands and exotic pastures of north-eastern, central and south-western Victoria. SLL are believed to have declined throughout their distribution and are currently known from only 10 conservation reserves across their entire distribution, there are however populations on roadsides and on private land. Major threats to the species include destruction and degradation of grassland habitat and the effects of fragmentation. High intensity grazing, inappropriate use of herbicide, rock collection or destruction, the spread of exotic grasses and other weeds, predation by cats and foxes and inappropriate fire regimes are all threats to the ongoing persistence of the species.

Within the MSA management area, the Western Grassland Reserve is the focus of monitoring and management for the species. While SLL is known to occur within the reserve, its extent and abundance is unknown. Management of the reserve focuses on biomass control including weed control via fire or grazing. SLL has a complex relationship with biomass. At very low biomass conditions, individuals may be exposed to increased predation, while excessively high biomass conditions can impact the species ability to thermo-regulate. Furthermore, they seem to benefit from high grass structural complexity which can be maintained by low-moderate intensity grazing (Howland et al. 2016). Individuals are also killed by fire and trampled by stock if biomass reduction occurs when lizards are active. Guidance on how best to manage the accumulation of biomass in the grassland reserve to promote persistence of the species is needed.

A PVA model for SLL exists that has explored options for translocation (O'Shea 2013, 2016) but this model did not explicitly investigate biomass management options. The model of O'Shea (2013), along with expert judgements elicited in a formal elicitation process were used to develop a PVA model to explore several alternative biomass control scenarios entailing different grazing and burning conditions. Insights from this modelling process can be used to identify important knowledge gaps, guide future data collection, and refine our understanding of the population dynamics of the species and potentially improve management of the species over time.

9.2 Methods

9.2.1 Conceptual models

Two conceptual models for the SLL were developed in collaboration with species experts. Firstly, a conceptual model of the life cycle of the species was developed to illustrate how threats and management influence important life stages (Figure 9.1). The species was described in two main stages, juveniles and adults. Adults are the reproductive stage and the rate of recruitment is influenced by the proportion of females mating, the number of eggs produced per reproductive female per year and the survival rate of hatchlings to become juvenile lizards. Threats include mortality due to fire and trampling by stock and biomass accumulation. Management levers include biomass control through grazing and fire regimes. These activities can also be a threat if not managed appropriately (Figure 9.1). Predation by birds is also a threat but control is more difficult to manage. We have not considered managing predation in this version of the model.

A second conceptual model illustrates the timeline for when SLL is active and when biomass control can have a negative impact on the species. SLL breeds between August and December and this is the time when animals are active. Fire during this time can increase mortality. Biomass control methods are less likely

to increase immediate mortality if scheduled between January and August. Fire as a biomass control method is thought to be best when done between January and April when the lizards can seek refuge in soil cracks (M. O'Shea, pers. comm.)(Figure 9.2).
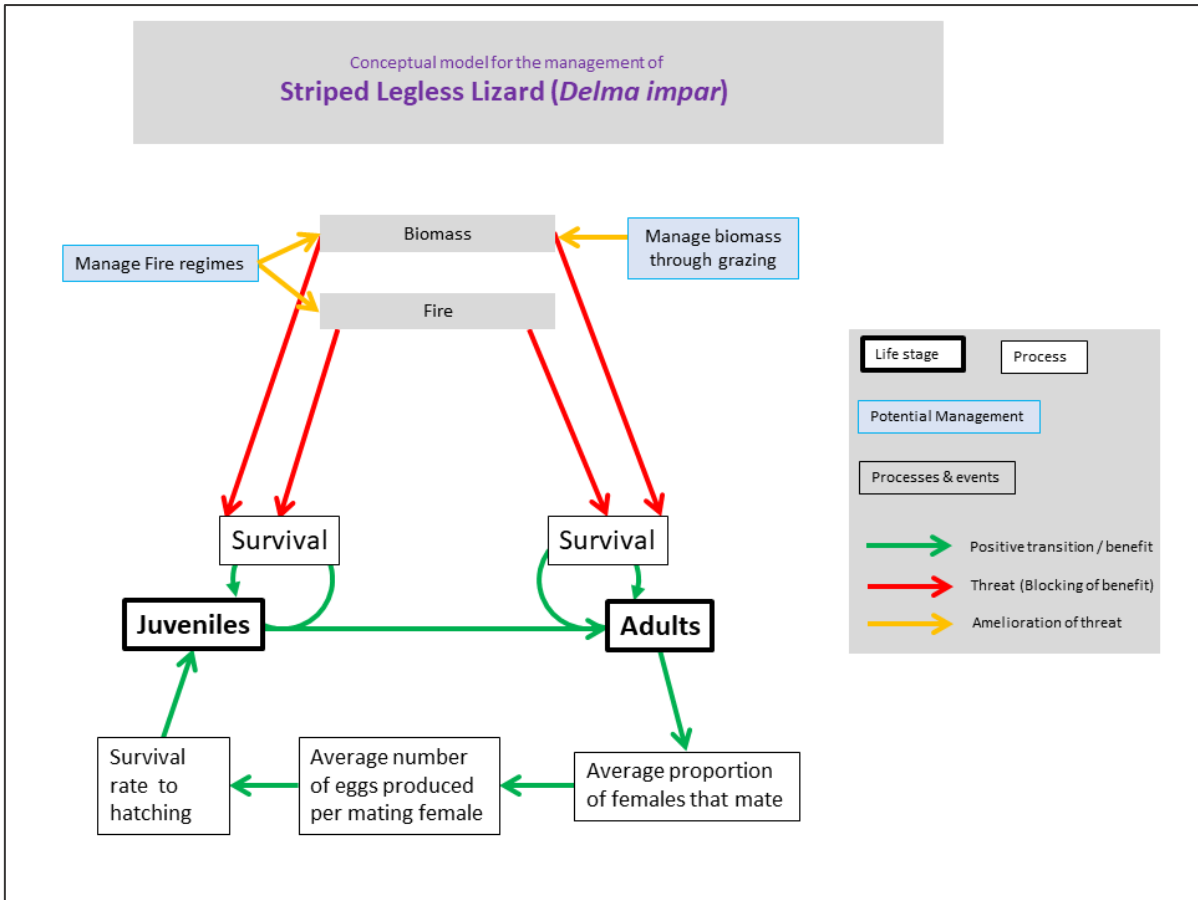

Figure 9.1: Conceptual model of the life cycle of Striped Legless Lizard illustrating the influence of threats and management.
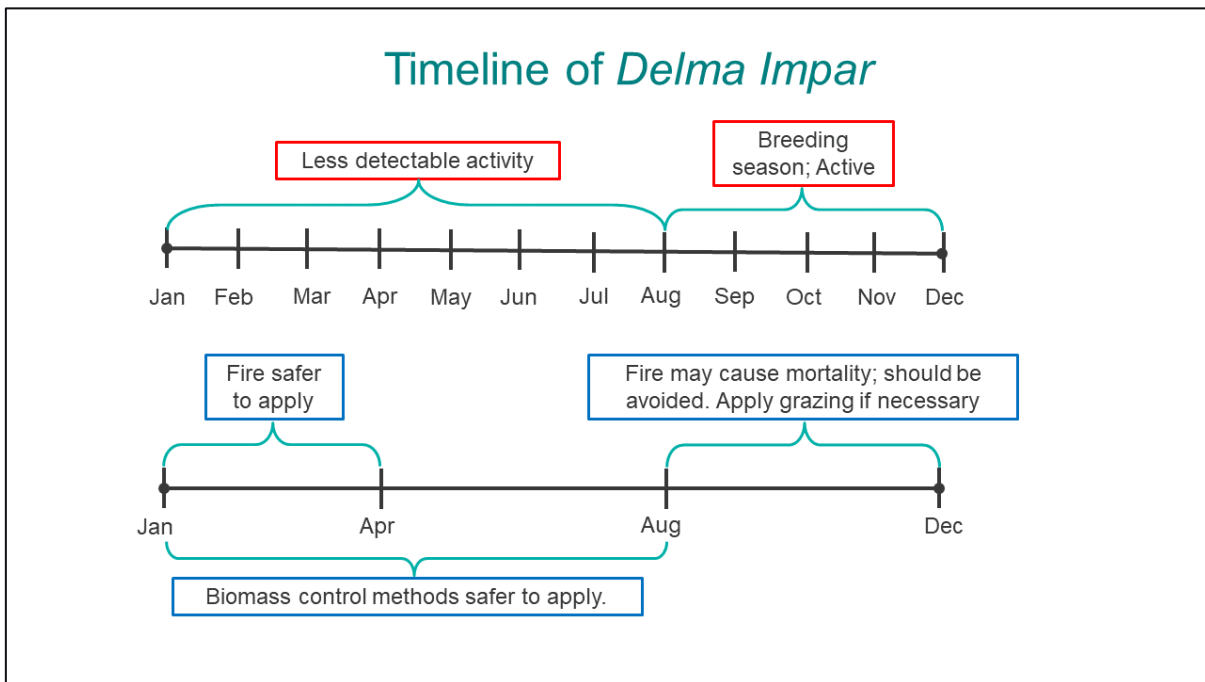

Figure 9.2: Timeline of when Striped Legless Lizard is active and the effect of biomass control.

9.2.2 Vital rates

As described above, a PVA model was developed for SLL using 2 stage classes, Juveniles and Adults. Fecundity of adults was estimated using expert judgement and verified through estimates presented in O'Shea (2013, 2016). The model assumes a pre-reproductive census such that the estimate of fecundity includes the sex ratio, the proportion of adult females reproducing per year, the number of clutches per female per year, the number of eggs produced per clutch and the survival of hatchlings to become juvenile lizards by the next census. Survival rates of hatchlings, juveniles and adults are influenced by the amount of biomass at the site which we assume is influenced by the time since the last biomass control event. Experts hypothesized that too little biomass exposes SLL to predation and reduces survival rates of all stages, while excessive biomass accumulation reduces survival rates due to the lack of exposed areas to thermo-regulate. This effect is also thought to depend on the type of biomass control. Fire tends to remove nearly all the biomass, putting SLL at higher risks of predation immediately post-fire, unlike grazing, which needn't remove all the biomass if the intensity and duration of grazing is limited). Monitoring data across the Western Grassland Reserve and several other conservation reserves within the MSA management area suggests that the rate of biomass accumulation is faster if fire is the control method compared to grazing (See Figure 1.1) (DELWP 2015).

Estimates of survival as a function of time since biomass control for fire and grazing were derived from expert judgements and data on biomass accumulation given fire and grazing. A panel of experts was asked to estimate survival under different percentages of vegetation cover for both fire and grazing biomass control methods. These judgements were then related to data on biomass accumulation, given time since biomass control for grazing and fire. Polynomial models were fit to these data to derive individual survival functions for hatchlings, juveniles and adults conditional on the two biomass control methods (Figure).

In the PVA model, mean survival rates each year changed depending on time since the last biomass control event according to the functions depicted in Figure 9.3. When a biomass control event occurs, then the survival rates are reset to the values of these functions when time since biomass equals zero.
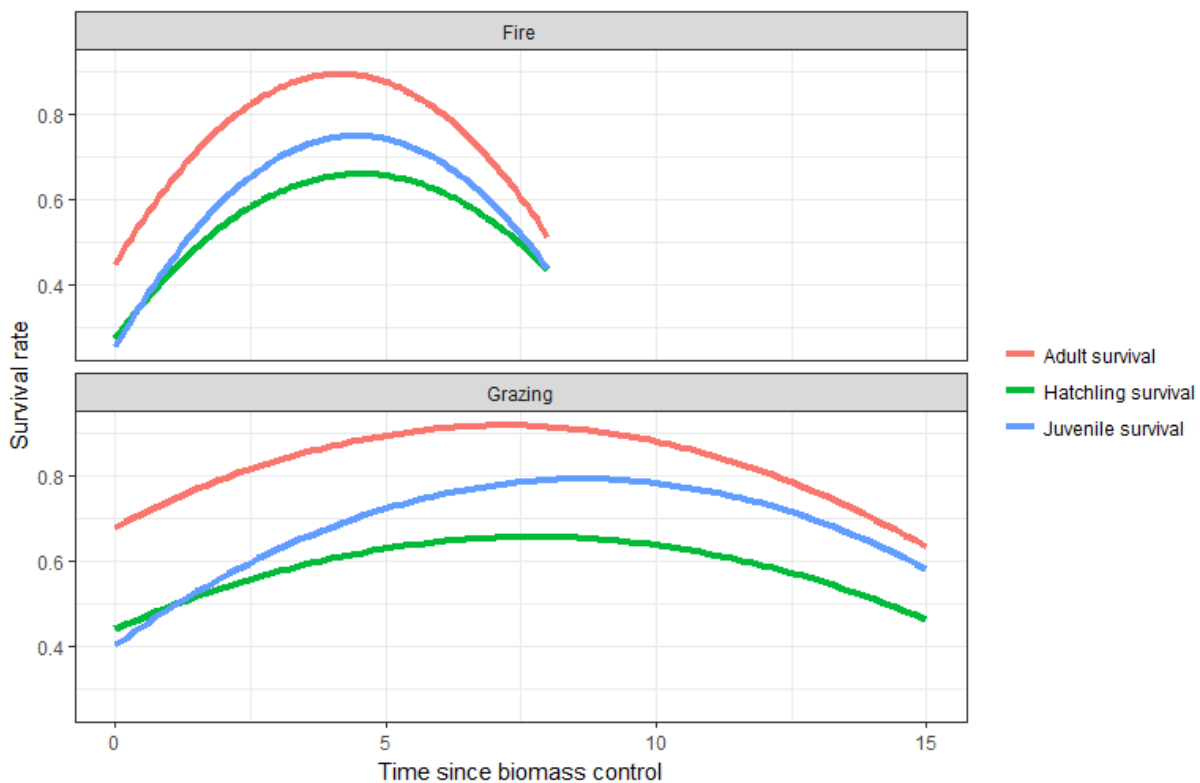

Figure 9.3: Expert-elicited survival rates for adult, juvenile and hatchling Striped Legless lizards as a function of time since biomass control when using fire (top) and grazing (bottom) as the method of biomass control.

9.2.3 Stochasticity

The level of natural variation in survival and fecundity is unknown and was not elicited. Instead we used default values that were assumed for all models where standard deviations could not be estimated from data. For fecundity estimates, standard deviations were derived based on a coefficient of variation (CV) of the mean vital rates of 20%. For survival rates, a CV of 10% was used. This assumption was evaluated as part of the sensitivity analysis. As data on variability in vital rates comes to hand, improved estimates can be incorporated into the model.

9.2.4 Additional model specification

Density dependence was incorporated as a ceiling model. In this type of model, populations can grow exponentially according to the vital rates until they reach a ceiling, and then remain at that level. Carrying capacities were set high at 10,000 individuals so that the ceilings could not be reached within timespan of the model. Instead populations were regulated through the relationship between vital rates and biomass accumulation and control.

Abundances within the Western Grassland Reserve are unknown. We set the initial abundance at 1000 individuals and assumed a stable age distribution. The choice of this abundance ensured that it was large enough to explore the benefits of alternative management scenarios and small enough to explore the influence of the alternative scenarios on extinction risk of the species. The model was run over 50 years for 1000 iterations. Model scenarios were compared using expected minimum abundance (EMA).

9.2.5 Management scenarios

A number of management actions to control biomass either through fire or grazing were explored (Table 9.1 and 9.2). Five fire scenarios were explored ranging across different combinations of fire interval and season as well as a no-fire scenario. Grazing scenarios assumed a medium intensity of grazing which is equivalent to 2.3 dry sheep equivalents (DSE) per hectare. Scenarios ranged from grazing every year to every 2, 3, 4 years as well as a no grazing scenario.

Impacts of fire and grazing might induce an immediate mortality when they occur. Experts estimate that spring burns are likely to have higher mortality than Autumn burns as lizards are more active in Spring, and soil crack refugia might be less available to lizards during Spring. Fire is likely to have higher immediate mortality than grazing irrespective of the season (Table 9.1).

Biomass accumulates at different rates post-fire in comparison to post-grazing, and the impact of biomass accumulation on SLL is different given the control method. These assumptions and the expert elicitation of survival rates resulted in very different temporal trends in survival for SLL after fire versus grazing, for long periods after biomass control. Given this difference in temporal patterns, we have assumed that for the fire scenarios the do nothing action assumes a prior history of controlled burning. For the grazing scenarios, the do nothing action assumes a grazing history. Combined fire/grazing scenarios have not been explored in this study. Management actions focusing on reducing predation pressure were not investigated explicitly in this study, but insights into the likely benefit of reducing predation pressure can be made from the results of the sensitivity analysis.

Table 9.1: Summary of fire management scenarios

Model number	Fire interval	Fire season	Immediate mortality (%)	Description
F1	short	either	13.75%	No burning 1 year after biomass control. Probability of burning 0.5 every year after that. Season is either Spring or Autumn
F2	medium	autumn	17.5%	No burning for 2 years after biomass control. Probability of burning 0.33 every year after that. Season is Autumn
F3	medium	spring	25%	No burning for 2 years after

			biomass control. Probability of burning 0.33 every year after that. Season is Autumn
F4	long	either	21.25%	No burning for 3 years after biomass control. Probability of burning 0.2 every year after that. Season is Autumn or Spring
F5	no fire	no fire	0.0	No biomass control but a history of fire

Table 9.2: Summary of grazing management actions explored with the PVA model.

Model number	Grazing frequency	Immediate mortality from trampling	Intensity
G1	Every year	8.25%	Medium intensity at 2.3 DSE ha^{-1}.
G2	Every 2 years	8.25%	Medium intensity at 2.3 DSE ha^{-1}.
G3	Every 3 years	8.25%	Medium intensity at 2.3 DSE ha^{-1}.
G4	Every 4 years	8.25%	Medium intensity at 2.3 DSE ha^{-1}.
G5	Every 5 years	8.25%	Medium intensity at 2.3 DSE ha^{-1}.
G6	No grazing	8.25%	No grazing but a history of grazing

9.2.6 Sensitivity analysis

Three types of sensitivity analysis were investigated: deterministic, stochastic and scenario sensitivity. The deterministic sensitivity analysis focused on the elasticities and sensitivities of the stage matrix which equate to the vital rates under the low biomass conditions. Elasticities reflect the proportional contribution of each vital rate to the growth rate under low biomass conditions. Sensitivities are indices of the influence of small changes in vital rates on the population growth rate. The stochastic sensitivity analysis involved adjusting each of the vital rates and standard deviations of the vital rates by ±10% each to investigate if the stochastic model was sensitive the values of the parameters. If the EMA results changed by more than ±10% this was taken to indicate that the model was sensitive to the value of the relevant parameter. Adult survival was altered by -10% and +5% to avoid survival rates >1.0.

The scenario sensitivity analysis evaluated the effect of varying the relationships between survival and time since biomass control that were elicited from the species experts for hatchlings, juveniles and adults for both fire and grazing scenarios (Figure 9.3). Each curve was adjusted by ±10% and applied to all management alternatives to determine whether the rank order of fire and grazing management scenarios change compared the base scenarios using the unchanged survival functions.

9.3 Results

9.3.1 Management actions

Of the management scenarios investigated, the results suggest that if fire is the biomass control method then a long fire interval in either season is the preferred option to enhance persistence of SLL (Figure 9.4). The model suggests that no burning or too frequent burning is detrimental to the species with both scenarios resulting in local extinction within 50 years. Medium fire interval in Autumn was better than in Spring but the benefits of were not large suggesting a longer interval between burning is more beneficial than season.

If grazing is the biomass control, then longer intervals between grazing is preferred with grazing every 5 years resulting in the highest EMAs of the scenarios tested (Figure 9.4). Grazing every year was the worst grazing option resulting in local extinction within 50 years. Grazing every 2 years was also detrimental and

the model suggests that frequent grazing is worse than doing nothing if the area has a history of grazing. Grazing is preferred over burning due to the high immediate mortality of SLL under a burning regime.
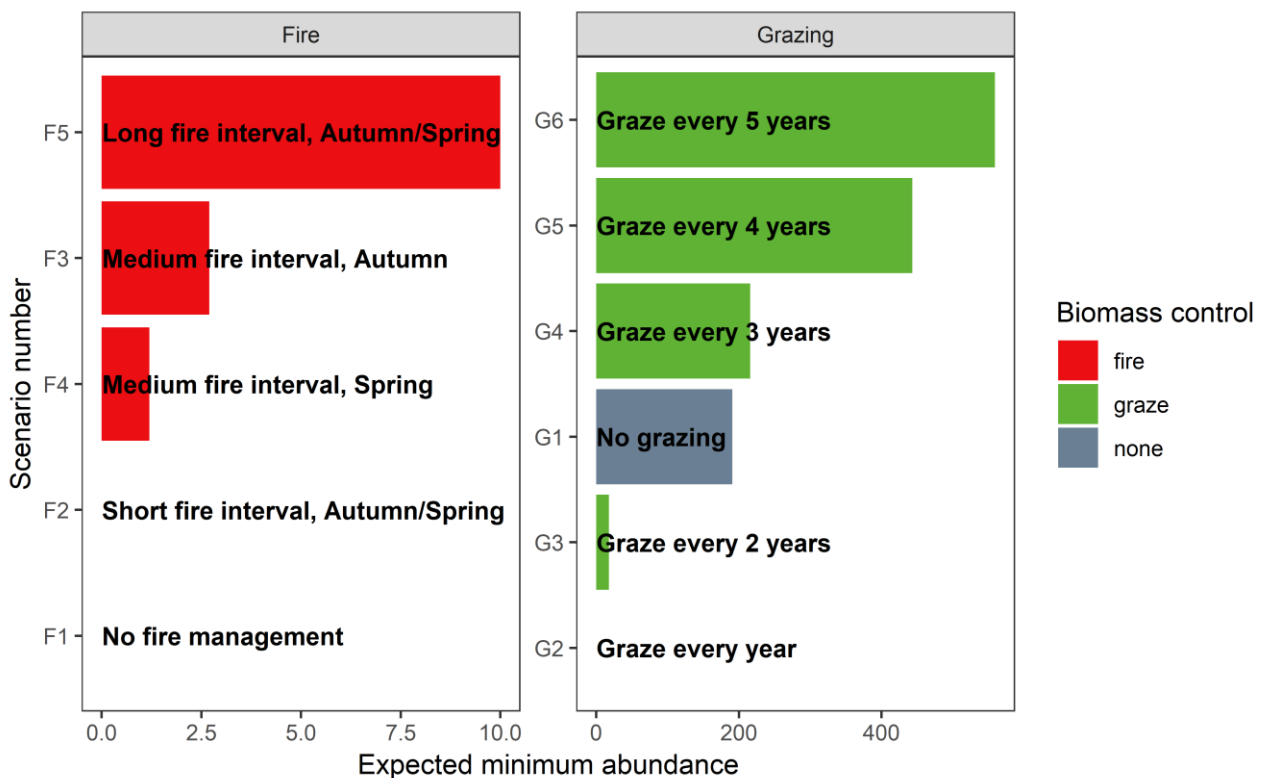

Figure 9.4: Ranking of management actions using expected minimum abundance for fire and grazing scenarios

9.3.2 Sensitivity analysis

Deterministic sensitivity analysis of the stage matrix revealed that adult survival is having the largest influence on the growth rate. Fecundity and juvenile survival have lower but equal influence on the growth rate. This is also reflected in the sensitivities where the growth rate is more sensitive to adult survival than juvenile survival and fecundity (Figure 9.5).

The stochastic sensitivity analysis revealed the model is sensitive to all vital rates in the model (Figure 9.6). This means that small changes in survival and fecundity can results in large changes in EMA. This will have implications when interpreting model results for informing best management for the species. While predator control was not formally investigated in this study, the results from the sensitivity analysis suggest that even small improvements to survival rates of SLL can have a large influence on its persistence into the future.

The scenario sensitivity revealed the robustness of the management actions ranks was variable depending on the change induced (Table 9.3 and 9.4). For instance, for the fire scenarios the ranks of actions were robust to increases in survival rates. However, when survival rates were reduced by 10% all models went to extinction so could not be ranked. For the grazing scenarios, the ranks were robust when survival rates decreased by 10% resulting in a rank correlation of 0.99 between the base scenarios and the altered scenarios. However, when survival rates were increased by 10% the rank correlation was only 0.66. This was influenced predominantly by the do nothing action. It ranked 4 in the base models and then ranked 1 when survival rates were increased by 10%.
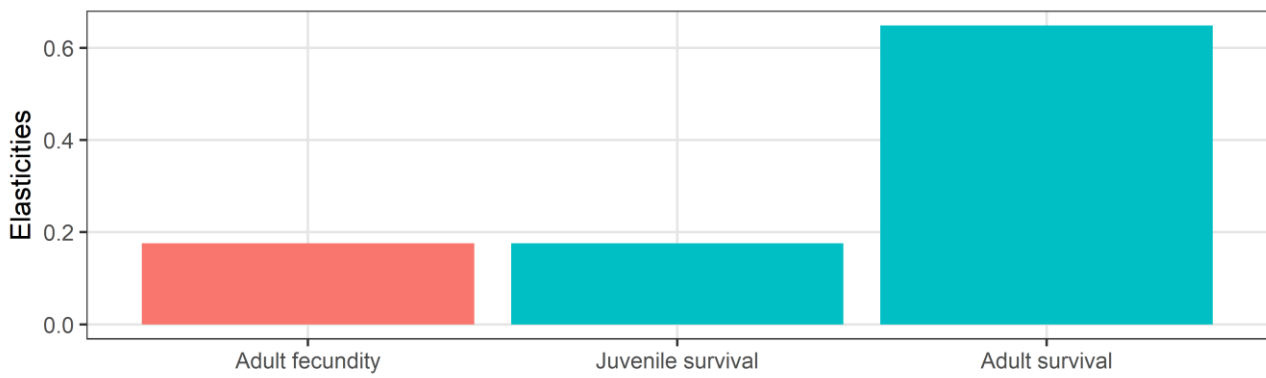
Figure 9.5: Elasticities of Striped legless lizard vital rates under low biomass conditions. Orange bar refers to fecundity estimates, and green bar refers to survival rates.
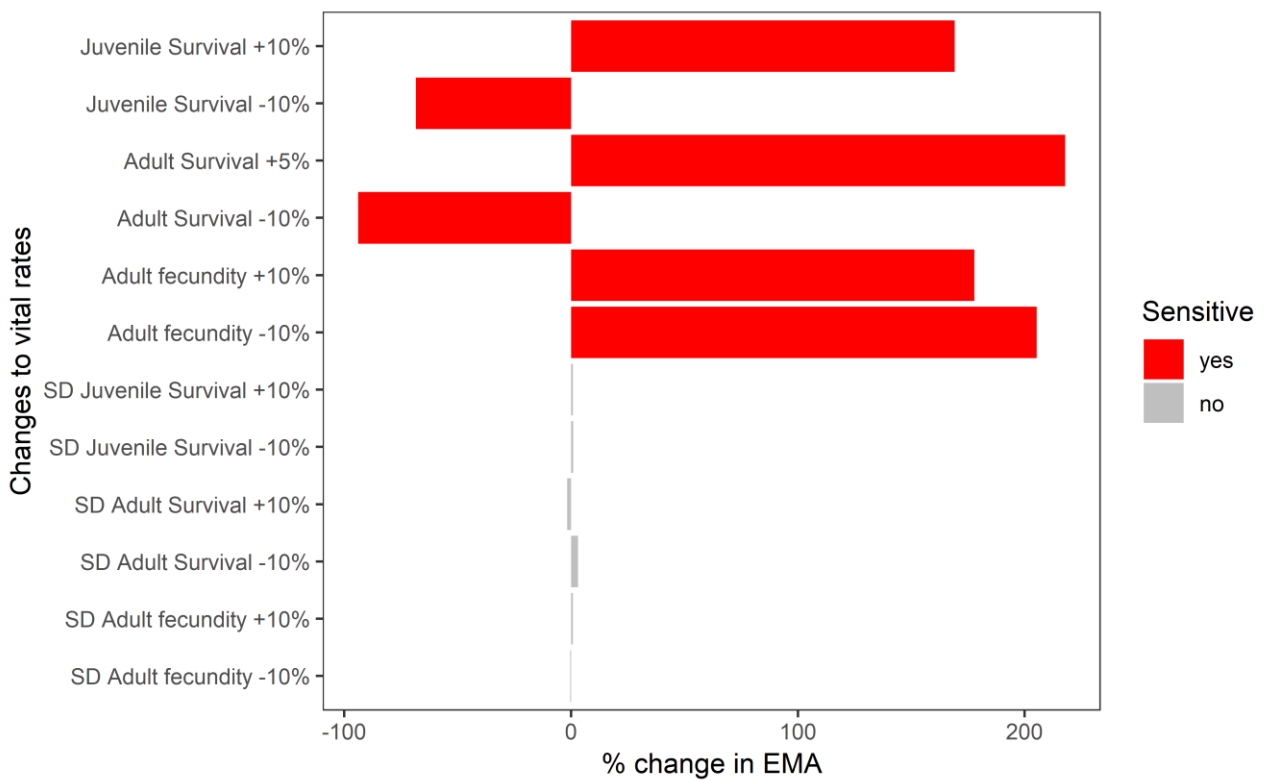

Figure 9.6: Stochastic sensitivity analysis for key model parameters measured as the % change in expected minimum abundance from the base model

Unofficial

Table 9.3: Scenario sensitivity analysis for fire management scenarios with ranks of actions given +- 10% changes in survival functions. Associated EMAs in parentheses

Model number	Base model	Survival function +10%	Survival function -10%
	Rank	Rank	Rank
F1	4.5 (0)	4 (1)	3.5 (0)
F2	2 (3)	2 (156)	3.5 (0)
F3	3 (2)	3 (101)	3.5 (0)
F4	1 (10)	1 (334)	3.5 (0)
F5	4.5 (0)	5 (0)	3.5 (0)
Correlation		0.97	NA

Table 9.4: Scenario sensitivity analysis for grazing management scenarios with ranks of actions given +- 10% changes in survival functions. Associated EMAs in parentheses

Model number	Base model	Survival function +10%	Survival function -10%
	Rank	Rank	Rank
G1	6 (0)	6 (0)	5.5 (0)
G2	5 (18)	5 (229)	5.5 (0)
G3	3 (216)	4 (656)	3 (5)
G4	2 (443)	3 (769)	2 (23)
G5	1 (559)	2 (806)	1 (53)
G6	4 (191)	1 (909)	4 (1)
Correlation		0.66	0.99

9.4   Discussion

The PVA model developed for SLL explored a range of management scenarios for two biomass control methods: fire and grazing. Existing data, models and the collective judgements of experts was used to construct and parameterize the model. The management actions explored were the same as for the other EPBC species present in the western grassland reserve, and which are covered by the MSA agreements. There may be other management scenarios that are optimal for SLL that were not explored here. Undertaking no management of biomass was found to be detrimental to the likelihood of persistence of SLL at the WGR. Of the management actions considered, our results suggest the preferred management option is grazing with long intervals ≥ five years apart. If fire is used to manage biomass, then long intervals ≥ five years, preferably in Autumn are preferred. Increased mortality due to fire and then increased predation due to less cover, may place SLL at higher risk of extinction immediately after fires compared to grazing. Biomass was found to accumulate much faster post-burning regime than was the case after the imposition of grazing so any negative effects of a transitory period of low biomass are likely to be greater when fire is used to control biomass compared to scenarios when grazing is used to control biomass. While this study did not investigate scenarios of predator control the model's sensitivity analysis suggest that even small improvements to survival can have large benefits to population persistence. Predator control scenarios should be developed further to explore this question in more detail.

Results regarding the benefits of grazing should be interpreted carefully and investigated more thoroughly before being used to inform on-ground management at the WGR or elsewhere in the species range. Grazing scenarios assumed a medium intensity of 2.3 dry stock equivalents (DSE) per hectare but did not specify the duration or whether an area is grazed by cattle or sheep. In addition, only one level grazing intensity was explored in this study. Before adopting a grazing regime for management of SLL, further investigation into the effects of grazing are needed. This should involve a more nuanced investigation into different grazing intensities, length of time an area is grazed (i.e. pulse grazing or longer-term grazing) and types of grazer (i.e. sheep or cattle). There is some evidence to suggest that a low-moderate intensity grazing regime is beneficial for *D. impar* (Howland et al. 2016).

Management scenarios explored in this study investigated fire or grazing. We did not investigate a combination of fire and grazing. Recent work on SLL in other parts of Victoria suggests there is a negative interaction between fire and grazing sites with very high fire and grazing frequencies having an elevated risk of extinction. This suggests a combined management regime of fire and grazing should be avoided as it has synergistic negative effects on the species (Scroggie et al. 2019).

The results of this study were largely influenced by SLL's relationship with biomass accumulation, suggesting the appropriate management of biomass is critical to the ongoing persistence of the species. However, our understanding of the rate at which biomass accumulates is uncertain and whether time since biomass control is the best measure of biomass accumulation. While there seems to be clear relationship between biomass accumulation and time since biomass control, other factors may influence biomass accumulation such as vegetation type, grassland composition, proportion of weeds in the system (i.e. especially Phalaris), biomass control method and other environmental conditions. Given this has a major influence on model outcomes, this is a critical knowledge gap that will apply not only to SLL but to all the species in this grassland system that are impacted by biomass accumulation.

The model is sensitive to all vital rates. This was evident in the deterministic, stochastic and to some degree the scenario sensitivity analysis. This suggests we have less confidence in model outputs and we need to be careful and manage experimentally rather than assuming the model is true. Despite this the model and results can be used to inform research on critical uncertainties and knowledge gaps. Estimates of fecundity were consistent across experts and supported by other studies (O'Shea 2013, 2016). The model was most sensitive to survival rates and the relationship between survival and biomass accumulation under different control methods. Improving knowledge on survival rates of this species under different conditions is a knowledge gap that is critical to better understand to ensure management is best tailored to support and promote persistence of SLL within the MSA management area. Curves that represent changes in vital rates as a function of time since biomass control were elicited from experts. Our sensitivity analysis only investigated shifts in the curve up and down by 10%. It is plausible the shape and steepness of the curves may be different and further sensitivity analysis is required to explore the influence of this assumption on model outputs. Lastly, further consultation with species experts and researchers is required to develop a research plan for SLL on how to best capture the data needed to estimate survival rates under different biomass conditions to improve model outputs and better inform management of the SLL into the future.

Unofficial

10 Growling Grass Frog (*Litoria raniformis*)

10.1 Background

The Growling Grass Frog (*Litoria raniformis*) is a large, and mainly aquatic frog species found throughout the south-eastern Australian mainland, and in Tasmania. Formerly common and widespread, this species is known to undergone significant population declines and local extinctions since the 1980s, though it remains geographically widespread, and can be locally abundant. Consequently, the Growling Grass Frog (GGF) is listed as *Vulnerable* under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999.*

Within the Melbourne Strategic Assessment Area, GGF often occurs close to permanent drainage lines such as rivers and streams, while also occurring elsewhere in the landscape wherever permanent or semi-permanent water is present. Commonly used off-stream habitats include farm dams, disused quarries and natural wetlands. GGF are known to readily utilise both artificial and natural waterbodies. Artificial wetlands constructed either specifically to serve as frog habitat, or as habitat for other wildlife are readily used. Artificial waterbodies constructed for other purposes such as storm water management or landscape amenity are also known to be occupied by GGF.

Within the MSA area, large areas of currently occupied habitat with either known or likely occurrences of Growling Grass Frogs are expected to become highly urbanised in the future and will therefore be rendered permanently unsuitable for continued occupancy. Several distinct zones of likely habitat, distributed along natural drainage lines (streams) have been identified for retention of GGF habitat in the MSA area (DEPI 2013). Compensation payments collected from developers who undertake urban development of specified areas of likely GGF habitat elsewhere within the MSA area are to be used to manage and enhance these retained habitat corridors. Creation of new, artificial wetlands within the retained habitat corridors has been identified as a major conservation action to be undertaken with funding derived from these compensation payments. Management of both existing and newly created wetlands to maintain and enhance habitat quality is also intended within the overall scope of the conservation works funded from compensation payments (DELWP 2017).

As creation of habitat has been identified as the major management response to habitat loss from urbanisation, it is necessary to determine preferred locations within the MSA GGF zones to place newly created wetlands to maximise the conservation benefits for GGF. Stochastic models for GGF metapopulations have been constructed (Heard et al. 2013, Rose et al. 2016), and can be used to assess alternative management scenarios for GGF in the Melbourne Strategic Assessment area.

10.2 Methods

10.2.1 Conceptual model

As there is currently no viable method for assessing the number of individual GGF inhabiting wetlands, a simpler modelling approach based on accounting for wetlands being occupied or unoccupied over time is used as both a conceptual and empirical basis for understanding the population biology of the GGF (i.e. a metapopulation model, (Heard and McCarthy 2012, Heard et al. 2013).

Stochastic patch-occupancy models (SPOM) for Growling Grass Frog have previously been developed based on a long-term monitoring dataset collected from populations on and around Merri Creek in the northern suburbs of Melbourne (2013). These long-term datasets document the processes of extinction and colonisation at wetlands within the Merri Creek corridor.

Conceptually, the model treats the annual probability of local population extinction as being a function of the size, connectivity to adjacent wetlands and habitat quality of wetlands, including two important dimensions of habitat quality: aquatic vegetation cover, and wetland permanence (hydroperiod).

The probability that vacant wetlands are colonised each year is treated as being dependent on connectivity to adjacent occupied wetlands. Connectivity is described by the distance-weighted sum of the number of

adjacent, occupied wetlands in the preceding timestep. Full details of the basic model are given in (Heard et al. 2013). The conceptual basis of the model is presented in Figure 10.1.
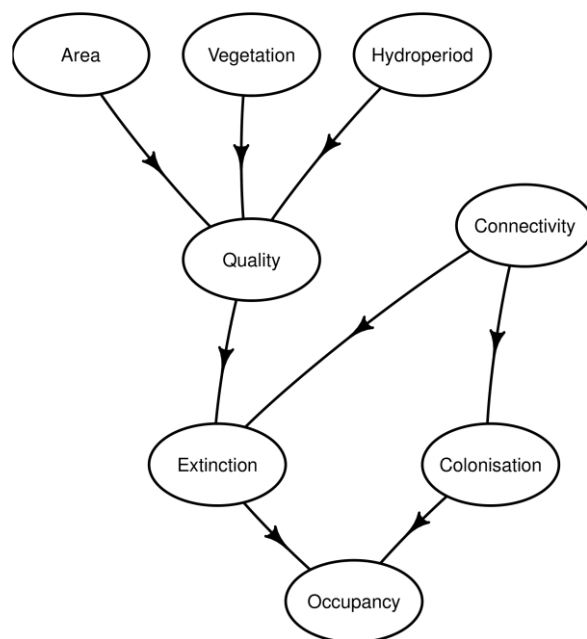

Figure 10.1: Conceptual relationship between wetland size, permanence, vegetation cover and rates of extinction and colonisation in the stochastic patch occupancy model (SPOM) for GGF metapopulations.

10.2.2 Model inputs

Rates of extinction and colonisation, conditional on the covariates (area, vegetation, permanence and connectivity) have been estimated from the repeated observations of occupancy at the Merri Creek wetlands using a Bayesian statistical model (Heard et al. 2013, Rose et al. 2016). As collection of occupancy monitoring data in the Merri Creek corridor is ongoing under MSA monitoring, the parameters of the SPOM can be updated from time to time using newly acquired occupancy data, and by adding further ecological data (e.g. new habitat variables, new ecological mechanisms, or improved understanding of the relationships between management actions and colonisation/extinction rates) to existing models ((Heard et al. 2013, Rose et al. 2016), Heard et al. in prep).

Annual extinction and colonisation events at each wetland are modelled using the Bernoulli distribution, with each wetland's extinction and colonisation probabilities being contingent on the prevailing habitat quality and connectivity variables. There is no explicit temporal variability in the vital rates of the model. Uncertainty in the values of the parameters of the model are incorporated into the model's predictions by estimating the joint probability distributions of the model's parameters, and then using multiple values sampled from these multivariate probability distributions for different runs of the stochastic model.

The parameters of the fitted model are used to simulate the performance of alternative management scenarios (see below) for GGF metapopulation in and around the GGF corridors across the entire MSA area. Many repeated runs of each scenario are used to estimate the relative conservation risks of alternative sets of management actions. The main management action considered was wetland construction, but in principle the model can be used to assess the effect of any management action that influences wetland quality or connectivity.

10.2.3 Management Scenarios

Based on expert opinion and previous observation of changes in stream hydrology and water quality in streams in heavily urbanised catchments in Melbourne, we assumed that in-stream habitats themselves would be of negligible habitat value once surrounding landscapes within the MSA were fully urbanised (Canessa and Parris 2013). Accordingly, alternative management scenarios have been assessed under the assumption that habitat for GGF would be provided solely by off-stream wetlands: both natural and artificial.

We considered scenarios where sufficient resources were available for the creation of a total of 50, 100 or 150 artificial wetlands across the entire network of GGF corridors in the MSA area. Many potential locations for artificial wetlands were identified by exhaustively searching the entire lengths of the corridors to determine the technical and logistic feasibility of wetland construction. This process was based on the opinions of a panel of expert wetland ecologists and hydrologists. A formal optimisation process (see below) was used to select preferred schemes for wetland creation from these many options, subject to the constraints of the three total wetland budgets (50, 100 and 150 wetlands).

As the impact of linear infrastructure such as roads and railways on dispersal and colonisation by GGF was uncertain, alternative management scenarios were considered where linear infrastructure formed either moderate (equivalent to frogs needing to disperse an extra 250 m) or complete (no dispersal possible) barriers to dispersing frogs (relative to open, unurbanized habitat).

There is uncertainty as to the extent to which urbanisation will degrade existing wetlands, as well as the extent to which artificial wetlands can replicate the attributes of high-quality existing wetlands. Therefore, we considered alternative management scenarios where permanence of wetland post-urbanisation was either high or low, and aquatic vegetation cover was also either high or low. These wetland quality attributes are both known to be important drivers of overall habitat quality for GGF. Aquatic vegetation and hydroperiod are used here as surrogates for a broader range of unmodeled aspects of habitat quality for which relevant data are not generally unavailable. These unmodeled (and mostly unmeasured) wetland attributes include water quality, quality of adjacent terrestrial foraging habitat, and the presence of predatory fish.

10.2.4 Decision analysis

Multiple runs of the SPOM are used to estimate the quasi-extinction risk associated with any given habitat creation scenario. The measure of (inverse) risk was the mean number of wetlands that were occupied by GGF after 50 years. As the number of possible combinations of wetlands to construct was extremely large (selecting 50, 100 or 150 wetlands to create from 300+ potential locations involves an intractably large number of options) it was not feasible to exhaustively evaluate the risks associated with every possible wetland creation scheme using the SPOM. As an alternative, a search of the overall decision-space was made using a stochastic optimisation algorithm (simulated annealing (Kirkpatrick et al. 1983)). Multiple optimisation runs were conducted to compare preferred allocations of wetland creation effort among corridors under alternative management scenarios (permanence, vegetation, dispersal).

10.3 Results

Optimised allocation of wetland construction effort amongst corridors under a variety of assumptions, and three proposed wetland construction budgets (50, 100, 150 wetlands) is shown in Figure 10.2. Regardless of the assumptions made, or the budget allocated, optimised solutions always involved allocation of a large proportion of wetland construction budget to the Merri Creek corridor, with somewhat smaller allocations to the Kororoit Creek and Southeast corridors. Under all scenarios and wetland creation budgets, the allocations of effort to the remaining corridors were always comparatively small.
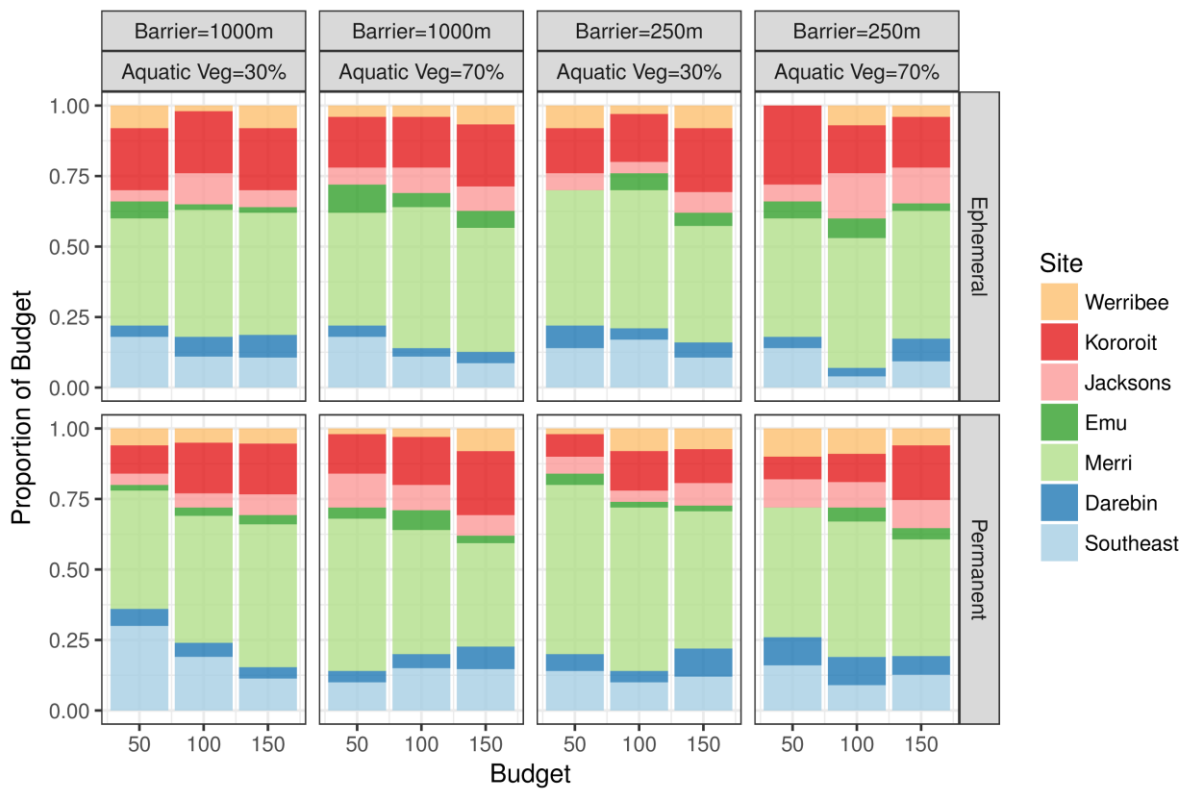
Figure 10.2: Optimised allocation of wetland creation effort among corridors for 3 budgets, 2 barrier levels, 2 aquatic vegetation levels, and 2 wetland permanence (hydroperiod) levels. Sizes of coloured bars represent the proportion of the total wetland construction budget to be spent in each corridor.

Risks associated with each scenario under each optimised allocation of wetland construction efforts among corridor are shown in Figure 10.3. The Merri Creek and Southeast corridors were predicted to have relatively low conservation risks, regardless of assumed wetland quality attributes or the strength of barrier effects. However, effects of increasing budget were clearly greatest in Merri Creek, with progressively greater numbers of constructed wetlands leading to a consistent reduction in risks. The impacts of barrier effects varied markedly between corridors, with the impacts of different assumptions about the strength of barrier effects having minimal impact on estimated risks in the Werribee and Kororoit Creek corridors, but much greater effects in the other corridors. This difference likely reflects the limited number of crossing structures (roads) that currently exist or are likely to be constructed as part of urban development of the Werribee and Kororoit corridors (DELWP 2017).

Risks of some corridors dropping to very low rates of occupancy during the management horizon (50 years) remain high, regardless of the management assumptions. In particular, the Werribee, Jacksons and Darebin corridors are all at risk of dropping below 25-30 occupied wetlands under several assumed scenarios, underlying the significant conservation risks inherent in managing these corridors. Conversely, the predicted risks of falling below 25-30 occupied wetlands for the Southeast and Merri corridors remain small, regardless of the assumptions about habitat quality or barrier effects.
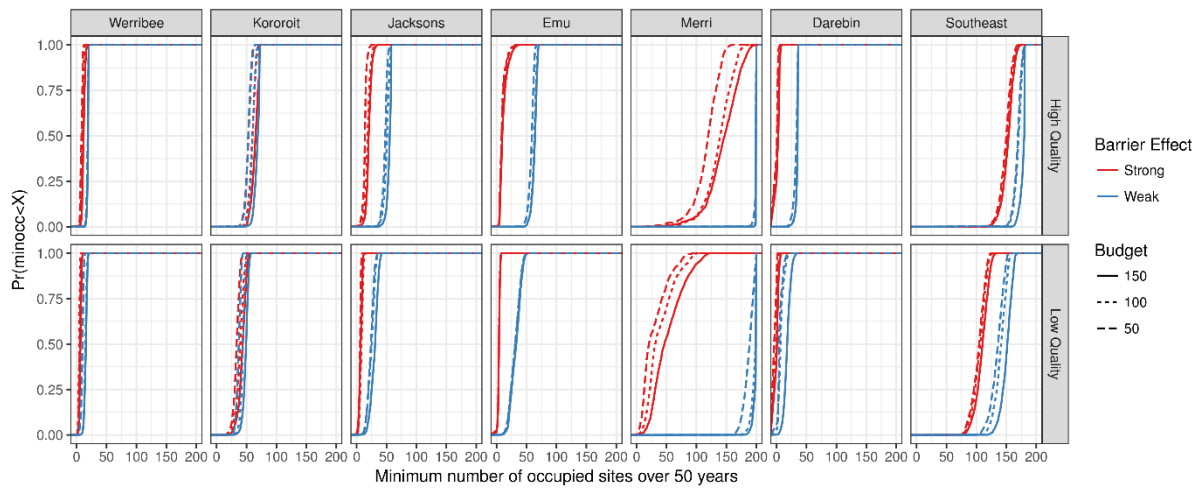
Figure 10.3. Cumulative probabilities (y-axis) of the number of wetlands in each corridor being occupied by GGF dropping below a given number (x-axis). Curves displace more to the right imply lower risk. "Low quality" implies wetlands with low vegetation cover and high permanence.

Predicted risks of the global number (i.e. all corridors combined) of wetlands occupied by GGF dropping below any given threshold (quasi-extinction) are given in Figure 4. The model predicts that even under the most pessimistic scenario, the number of occupied wetlands across the entire MSA is unlikely to drop below 150 over a period of 50 years. The strength of barrier effects has a large impact on estimated risks, while wetland creation budget and habitat quality also contributed to overall risks (Figure 10.4).

Figure 10.4. Global risks of quasi-extinction under several scenarios. "Low quality" implies wetlands with low vegetation cover and low permanence, "High quality" implies wetlands with high vegetation cover and high permanence.

10.4 Discussion

Our analysis provides clear, robust findings regarding the appropriate allocation of wetland creation effort among the different corridors. Regardless of assumptions regarding wetland quality, or the effects of infrastructure barriers on connectivity, optimised solutions always involved a high propor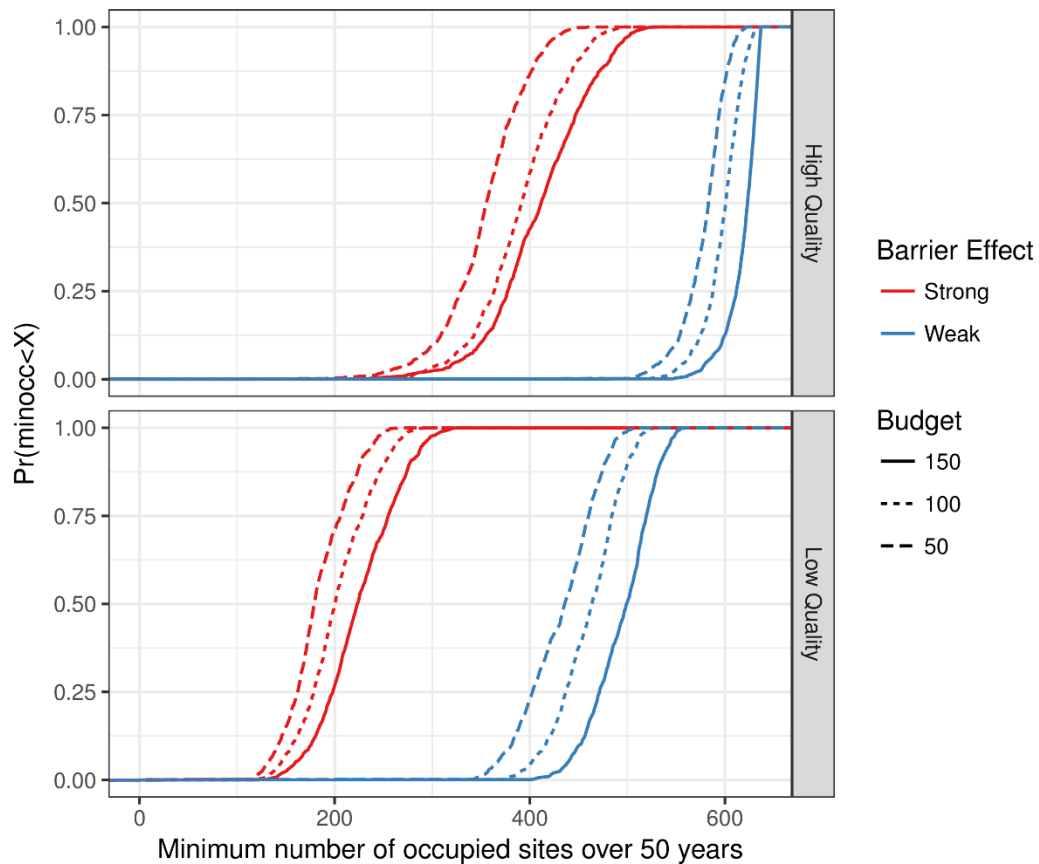tional allocation of wetland construction effort to the Merri, Kororoit and Southeast corridors, and smaller allocations to all other corridors. Regardless of the allocations or the assumptions, predicted conservation risks varied strongly amongst corridors (Figure 10.3).

Assumptions about barrier effects and habitat quality had much stronger effects on estimated risks in some corridors than others. For example, assumptions about the strength of barrier effects had minimal impact on estimated risks in the Werribee and Kororoit creek corridors, but much large effects on modelled risks in Merri and Emu creeks. These differences reflect the different extent to which corridors are predicted to be impacted by construction of linear infrastructure such as roads – in corridors where minimal construction is anticipated, the associated impacts on conservation risk will be small regardless of the strength of the barrier effect.

Estimates of conservation risks resulting from this analysis are almost certainly underestimates of true risk, as the model only accounts for a select few risk processes inherent in the ecology of the metapopulation systems. Rare, but high negative impact events (catastrophes) are not included in the model and could result in markedly higher realised risks than the model can predict (Beissinger and Westphal 1998, Brook 2000). Similarly, the SPOM does not include temporal variability in the extinction or colonisation rates due to effects such as climatic variation. Inclusion of such variability in the model would be expected to result in a greater spread of possible outcomes, including more trajectories that drop to very low rates of occupancy, or indeed extinction. The model also does not include genetic phenomena such as inbreeding depression in assessments of conservation risks. Given that the corridors will be demographically isolated from one another, and will likely have reduced connectivity to populations in adjacent habitats outside the MSA area, risks of heightened extinction risk due to inbreeding depression are likely to be significant over the longer term (Hale et al. 2013, Keely et al. 2015).

The parameters of the metapopulation model are derived solely from data collected along the Merri creek corridor and surrounding areas. It is implicitly assumed that the resulting stochastic model is a reasonable description of the dynamics of GGF metapopulations elsewhere in the MSA area. Similarly, much of the data that underpins the SPOM is collected from landscapes with lower levels of urbanisation (although some data from more heavily urbanised landscapes is included as well). Predicting the behaviour of the same metapopulations once the surrounding landscapes are heavily urbanised involves a level of extrapolation and assumption that should be checked with observation of the dynamics of the metapopulation systems once urbanisation has occurred.

The approach to conceptualising wetland quality in the model is also very simple and only considers two aspects of quality (aquatic vegetation and wetland permanence/hydroperiod). Other aspects of wetland quality are not explicit in the model, though are captured implicitly in the decision analysis for prioritising wetland creation. Extension of the model to capture the influences of other aspects of wetland quality is a high priority. This could be done by collecting new data, but could also be informed by expert opinion, including formal expert elicitation (Hemming et al. 2018).

The existing models treats habitat quality as a static, unchanging feature of wetlands. Moreover, the models do not yet provide a means of linking management actions undertaken at the wetland scale to the resulting quality of the wetland, which is itself a dynamic quantity, dependant on a range of management practices, climate and the presence of other biota such as predatory fish and invasive weeds. As the intention is to undertake active management of both existing and newly created wetlands, ideally the model should provide predictive links between management actions, wetland habitat attributes, and the resulting wetland quality (measured as a probability of local extinction).

A range of wetland management practices are envisaged within the MSA area including eradication of predatory fish, control of weeds, management of surrounding terrestrial vegetation, active management of hydroperiod, active management of water quality and periodic de-silting. At present, the effectiveness of these actions on habitat quality as perceived by GGF is poorly known, so it is difficult to make effective

Unofficial

decisions regarding when and where to apply these actions to maximise conservation benefits. Expert opinion will be of some value here. The availability of a large number of wetlands, the intention to carry out management actions over a prolonged period of time, and the intention to monitor population outcomes for the GGF (DELWP 2015) provide an opportunity to implement adaptive management (*sensu* (Walters 1986)). By observing the outcomes of management actions over time, it will possible to progressively update and improve existing conceptual and stochastic models, with the intention of improving the effectiveness of management over time.

11 Management Recommendations for MSA species

In this project we developed population viability analysis (PVA) models for eight EBPC listed species occurring in the MSA area, with a separate modeling process undertaken for Growling Grass Frog. The motivation for developing the models was to provide an evidence base for decision-making for management of threatened species. The models integrate empirical data and expert judgement in a transparent, logical and consistent way to estimate the relative merits of alternative management actions towards the persistence of each species. This type of work will provide a strong foundation for on-going research to better inform our understanding of extinction risk of these species. It will also provide a valuable on-going resource to inform and improve management of the species over time and will make an important contribution to the MSA meeting its ecological commitments.

11.1 Summary of Results for MSA species

11.1.1 Grassland species

Accumulation of biomass over time is thought to be one of the biggest threats to long term persistence of the grassland species within the MSA. Biomass control can be managed through either planned burning or grazing but these activities have both positive and negative effects on the species. Planned burning can increase mortality in the year it occurs but also increases opportunities for germination and recruitment in low biomass conditions. Similarly grazing can reduce biomass and promote recruitment but several species are sensitive to grazing due to trampling and browsing. Thus, there is a trade-off between mortality and recruitment when choosing the form of biomass control and the timing, frequency and intensity with which it occurs.

Our PVA modelling indicates that all grassland species benefit from some form of biomass control. There were however differences in the preferred biomass control action between species (Table 11.1). This difference was most pronounced between the plants and animals, with Golden Sun Moth and Striped Legless Lizard preferring grazing regimes over burning regimes. These two species also perform better under different grazing intensities and frequencies. Amongst the plants there was either a preference for short fire intervals (SRF, BWW and LFG) or medium fire intervals (MFL and SGMO). These results suggest that when developing specific management plans for areas with overlapping species, trade-offs will be necessary.

For several species there were additional benefits of performing other actions that could be applied in conjunction with biomass control. Large-fruit Groundsel and Button Wrinklewort may benefit from supplemental watering to simulate a more frequent rainfall pulse. Matted Flax-lily and Small Golden Moths Orchid may benefit from supplemental watering during drought. Our models also showed a benefit to Small Golden Moths Orchid from excluding rabbits. Golden Sun Moth could benefit from the removal of perches (to reduce bird predation) and effective exclusion of cats. In all cases these actions provide benefits over and above ideal biomass management for MSA species.

Management actions explored were limited to biomass control and in some cases additional actions for watering and predator control. There may be other actions that promote persistence of species such as supplemental planting, translocation and genetic rescue. Weed control was assumed to be part of the biomass control actions and not modelled explicitly. However, different types of weed management would be explored to better understand the costs and benefits of different methods and intensities of weed management. Future version of the models could be adapted to explore different types of management actions.

Table 11.1: The top biomass control actions and other beneficial actions for grassland species.

Species	Top biomass control action(s)	Other beneficial actions modelled
Spiny Rice-flower	Short fire interval, Autumn/Spring	NA
Button Wrinklewort	Short fire interval, Autumn/Spring	Supplemental watering
Matted Flax-lily	Medium fire interval, Autumn	Watering during drought
Small Golden Moths	Medium fire interval, Autumn	Rabbit exclusion Watering during drought
Large-fruit Groundsel	Short fire interval, Autumn/Spring	Supplemental watering
Golden Sun Moth	High intensity annual grazing Short fire interval, Autumn/Spring (where grazing not feasible)	Remove perches Cat control
Striped Legless Lizard	Medium intensity grazing every 5 years Long interval fire, Autumn/Spring (where grazing not feasible)	NA

11.1.2 Non-grassland species

For Southern Brown Bandicoot the top ranked action was broad scale fox baiting (at 25% compliance) and the second highest ranked action was land acquisition and creation of SBB habitat connectivity network with targeted fox baiting (Table 11.2). Growling Grass Frog received the highest benefit from the allocation of the highest proportion of wetland construction effort in the Merri (overall highest allocation), Kororoit and Southeast corridors, implying relatively less construction effort be allocated in the Werribee, Jacksons, Emu and Darebin corridors (Table 11.2).

Table 11.2: The top actions for non-grassland species

Species	Top action	Other beneficial actions
Southern Brown Bandicoot	Broad scale fox baiting (25% compliance)	Land acquisition and creation of SBB habitat connectivity network with targeted fox baiting.
Growling Grass Frog	Highest proportion of wetland construction effort allocated to the Merri, Kororoit and Southeast corridors	

11.2 Summary of sensitivity analyses

Model sensitivity was explored through altering the vital rates in the models (stochastic sensitivity) and running management scenarios and assessing if the rank order of preferred actions changed with changes in underlying model assumptions (scenario sensitivity). Sensitivity analysis is important because it illustrates how robust models are to our assumptions and the uncertainty in the data used as inputs in the model. Thus, it has a direct influence on our understanding and confidence of the best action to promote persistence of species. It can also be used to guide the acquisition of new knowledge with the objective of reducing uncertainty in sensitive parameters.

Our sensitivity analysis revealed that all species models were sensitive to changes in the survival and fecundity rates (Table 11.3). Scenario sensitivity analysis revealed altering assumptions about the relationship between fire and vital rates (particularly survival and fecundity) can impact the ranking of actions

for all the MSA plant species. The results for several species were also sensitive to changes in the survival rates and the results for one species, Spiny Rice Flower, were sensitive to changing the initial abundance. The ranking of actions for Golden Sun Moth was not altered by any changes to assumptions. For most species the best action did not change under different assumptions regarding these parameters. The exceptions to this were Button Wrinklewort, Small Golden Moths Orchid and Striped Legless Lizard.

While the sensitivity analysis was informative, it was limited in the number of parameters we could explore. It is likely there are other assumptions and parameters the models are sensitive to, that were not investigated here. We recommend a further sensitivity analysis to more fully explore the behaviour of the models and how assumptions and uncertainty effect model outcomes. Furthermore, several of the models are completely or predominantly expert driven (e.g. Small Golden Moths Orchid and Golden Sun Moth). For such species, even though sensitivity analysis did not reveal the model was very sensitive to the parameters and assumptions we tested, if the experts were incorrect about inputs and assumptions then this could have major implications for our expectations of preferred management for the species. There is a clear imperative to gather basic life history data to confirm or refine the parameters estimates for the models of these species and to perform further sensitivity analysis to more comprehensively explore the model behaviour.

Table 11.3: Summary species' highest sensitivity to model parameters. The column "Stochastic sensitivity" lists all parameters that induced a change of >10% in EMA, the column "Scenario sensitivity" describes the changes to the models that caused the biggest rank order change in models (i.e. the lowest correlation coefficient)

Species	Stochastic sensitivity	Scenario sensitivity
Spiny Rice-flower	Survival ±10%	Decrease in initial abundance
	Germination ±10%	Survival vs TSF relationship
	Fecundity +10%	Decrease in survival rate
Button Wrinklewort	Survival ±10%	Decrease in survival rate
	Fecundity -10%	
Matted Flax-lily	Survival +1%	Survival vs TSF relationship
	Survival -10%	
	SD survival -10%	
	Fecundity ±10%	
Small Golden Moths Orchid	Survival ±10%	Fecundity vs TSF relationship
	Fecundity ±10%	
Large-fruit Groundsel	Survival ±10%	Survival vs TSF relationship
	Fecundity ±10%	
Golden Sun Moth	Survival ±10%	None
	SD survival +10%	
	Fecundity ±10%	
	Carrying capacity +10%	
Southern Brown Bandicoot	Survival ±10%	Increase in survival rate
	Fecundity ±10%	Decrease in survival rate
	Carrying capacity -10%	
Striped Legless Lizard	Juvenile survival ±10%	Increase in survival rate
	Adult survival +5% -10%	
	Fecundity ±10%	

Unofficial

11.3 Concluding remarks and next steps

We have constructed a set of population viability models for MSA species, this however should be regarded as the beginning of the process. These models should be embedded in an adaptive management cycle whereby the models are used to guide the acquisition of new knowledge and are then regularly updated with new information to improve subsequent management decisions.

Further refinements could be made by exploring a greater range of management options. The options assessed here were determined in consultation with land managers and species experts, and thus reflect what is believed to be most feasible given our current understanding of the species in the MSA context. Nevertheless, new management options will arise in the future and this will present an opportunity to further refine these models. For example, the current models ignore the role of genetics in species conservation. This was a deliberate decision given the amount of speculation required to include this in our models. This may be included in future iterations, particularly for species such as Button Wrinklewort which is the subject of a current research project investigating genetic rescue. It is plausible that new threats to MSA species will emerge in the future (e.g. disease) and this will require a reassessment of management priorities and may require an immediate response. The impact of new threats and the possible management levers could be investigated through updating these models.

Using these models to inform specific decisions requires accounting for other factors that may not be captured in the models. Other factors such as costs and feasibility of management; site specific constraints and whether species overlap and have different preferred management actions. There may also be other important values present that need to be accounted for (e.g. ecological communities and cultural heritage values). The models will also need to be updated and rerun so they reflect the initial conditions at the site/s in question.

Decisions in practice are likely to require trade-offs. In fact, our models revealed trade-offs may be necessary when choosing management actions for locations where multiple species reside. Trade-offs will largely depend on how decision makers view the difference between the expected benefits of alternative actions and how much they are willing to give up in benefits for one species for gains in other species or non-ecological values. The models developed here do not perform this task. Instead they provide estimates of the consequences of different actions for each species that can be used to explore trade-offs. The MSA program is currently piloting a structured decision-making approach (SDM) to explore how best to use these models in specific decision contexts and methods for exploring trade-offs (Machunter et al. 2019).

The PVA models developed in this study for the MSA program represent a wealth of data and expert knowledge. The intent is that models will be used to inform management and guide further research. We recognise the importance of communicating the model details and outputs with the people who are most likely to use them: the land managers and researchers. To facilitate this, we have developed a web-based tool which allows for the investigation of the PVA model results and comparisons between species and management scenarios (https://ari-eas.shinyapps.io/MSA_PVA/). We encourage users to explore the model outputs and provide feedback so that future iterations of the models can be improved and be informed by the most up to date information. We envisage that future versions of this tool will provide guidance on the appropriate use of model outputs for use in decision making, incorporate results of the SDM project and include the ability to explicitly explore trade-offs, so that managers and decision makers have a tool at their disposal that captures the current scientific evidence and methods for informing evidence-based decisions.

References

Akçakaya, H.R. (1991) A method for simulating demographic stochasticity. Ecological Modelling 54 (1-2), 133-136

Akçakaya, H. R. and Root, W. R. (2013) RAMAS GIS. Version 6.0. Applied Biomathematics, Setauket. New York

ARAZPA (1996) Population and Habitat Viability assessment (PHVA) for the striped legless lizard Delma impar. Australasian Regional Association of Zoological Parks and Aquaria, ACT Parks and Conservation Service, Striped Legless Lizard Working Group, and Conservation Breeding Specialist Group (SSC/IUCN).

Backhouse, G., and Lester, K. (2010) National Recovery Plan for the Small Golden Moths Orchid Diuris basaltica. Department of Sustaiability and Environment, Melbourne, Victoria.

Beissinger, S. R., and Westphal, M. I. (1998) On the Use of Demographic Models of Population Viability in Endangered Species Management. The Journal of Wildlife Management 62:821-841.

Belcher, R. O. (1983) New Australian species of Erechthitoid Senecio (Asteraceae). Muelleria 5:119-122.

Brook, B. W. 2000. Pessimistic and Optimistic Bias in Population Viability Analysis. Conservation Biology 14:564-566.

Broughton, S. K., and Dickman, C. R. (1991) The effect of supplementary food on home range of the Southern Brown Bandicoot, Isoodon obesulus (*Marsupialia, Peramelidae*). Australian Journal of Ecology 16:71-78.

Brown, G., Tolsma, A., and McNabb. E. (2012) Ecological aspects of new populations of the threatened Golden sun moth Synemon plana on the Victorian Volcanic Plains. Victorian Naturalist 129:77-85.

Brown, G. W., and Main, M. L. (2010) National Recovery Plan for the Southern Brown Bandicoot Isoodon obesulus obesulus Department of Sustainability and Environment, Victoria.

Burgman, M., Carr, A., Godden, L., Gregory, R., McBride, M., Flander, L. and Maguire, L. (2011) Redefining expertise and improving ecological judgment. Conservation Letters, 4, 81-87.

Canessa, S., and Parris. K.M. (2013) Multi-Scale, Direct and Indirect Effects of the Urban Stream Syndrome on Amphibian Communities in Streams. PloS one 8:e70262.

Carr, G. W., and Horsfall, P. F. (1995) Studies in Phormiaceae (Lilaceae) 1: New Species and Combinations in Dianella Lam. Ex Juss. Muelleria Australia Journal of Botany 8:365-378.

Carr, G. W., and Rodda, L. R. (2011) Matted Flax-lily (Dianella amoena) Translocation: Monitoring Spring/Summer 2010-2011. Ecology Australia, Fairfiled, Victoria.

Carter, O. (2010) National Recovery Plan for the Matted Flax-lily Dianella amoena. Department of Sustainability and Environment, Victoria.

Carter, O., and Walsh, N. (2006) National Recovery Plan for the Spiny Rice-flower Pimelea spinescens subsp. spinescens. Department of Sustainability and Environment, Melbourne, Victoira.

Caswell, H. (2001) Matrix population modes: construction, analysis and interpretation (2nd ed.). Sinauer Associates, Sunderland, MA

Courchamp, F, Berek, L. & Gascoinge, J. (2008) Allee Effects in Ecology and Conservation, Oxford University Press, Oxford, UK.

Cutten, J., and Squire, E. (2003) Messent Conservation Park, 2002 post-fire survey of Senecio macrocarpus and Thelymitra epipactoides. Department of Water, Land and Biodiversity Conservation, Prepared for South East Region National Parks and Wildlife, South Australia.

DELWP (2015) Monitoring and Reporting Framework: Technical Protocols for Program Outcomes. Department of Environment, Land, Water and Planning, East Melbourne.

DELWP (2017) Growling Grass Frog Masterplan for Melbourne's Growth Corridors. Department of Environment, Land, Water and Planning, East Melbourne.

Department of the Environment (2012) Synemon plana — Golden Sun Moth. Department of the Environment, Canberra, ACT.

Unofficial

DEPI (2013) Biodiversity Conservation Strategy for Melbourne's Growth Corridors. Department of Environment and Primary Industries, Melbourne, Victoria.

DPCD (2009) Delivering Melbourne's newest sustainable communities. Program Report. Department of Planning and Community Development, East Melbourne, Victoria.

DSE (2004) Golden Sun Moth Synemon plana. Department of Sustainability and Environment, East Melbounre, Victoria.

DSE (2009) Delivering Melbourne's newest sustainable communities. Strategic impact assessment report for the Environmental Protection and Biodiversity Conservation Act 1999. Department of Sustainability and Environment, East Melbourne, Victoria.

DSE (2013) Advisory List of Threatened Vertebrate Fauna in Victoria - 2013. Department of Sustainability and Environment, East Melbourne, Victoria.

Eldridge, D. J., Poore, A. G. B., Ruiz-Colmenero, M., Letnic, M. and Soliveres, S. (2016) Ecosystem structure, function, and composition in rangelands are negatively affected by livestock grazing. Ecological Applications 26:1273-1283.

Fordham, D.A., Akçakaya, H.R., Araújo, M.B., Elith, J., Keith, D.A., Pearson, R., Auld, T.D., Mellin, C., Morgan, J.W., Regan, T.J., Tozer, M. Watts, M.J., White, M., Wintle, B.A., Yates, C., and Brook, B.W. (2012) Plant extinction risk under climate change: are forecast range shifts alone a good indicator of species vulnerability to global warming? Global Change Biology, 18: 1357-1371

Gibson, L., and New, T. R. (2007) Problems in studying populations of the golden sun-moth, Synemon plana (*Lepidoptera: Castniidae*), in south eastern Australia. Journal of Insect Conservation 11:309-313.

Gill, J., Sethi, N., Mohan, A., Datta, S. and Girdhar, M. (2018) Glyphosate toxicity for animals. Environmental Chemistry Letters 16:401-426.

Hale, J. M., Heard, G.W., Smith, K.L., Parris, K.M., Austin, J.J., Kearney, M. and Melville, J.(2013) Structure and fragmentation of growling grass frog metapopulations. Conservation Genetics 14:313-322.

Heard, G., and McCarthy, M.A. (2012) Metapopulation viability of the Growling Grass Frog in Melbounre's urban growth areas. Report prepared for Depapartment of Sustainability and Environment. School of Botany, University of Melbourne.

Heard, G. W., McCarthy, M. A., Scroggie, M. P., Baumgartner, J. B. and Parris, K.M. (2013) Bayesian models of metapopulation viability, with application to an endangered amphibian. Diversity and Distributions 19:555-566.

Hemming, V., Burgman, M. A., Hanea, A. M., McBride, M. F. and Wintle. B. (2018) A practical guide to structured expert elicitation using the IDEA protocol. Methods in Ecology and Evolution 9:169-180.

Hills, A., and Boekel, R. (1996) Action statement No. 68. Large-fruit groundsel, *Senecio macrocarpus*. Department of Natural Resources and Environment, Victoria.

Howland, B. W. A., Stojanovic, D., Gordon, I. J., Fletcher, D., Snape, M., Stirnemann, I. A., and Lindenmayer, D. B. (2016) Habitat preference of the striped legless lizard: Implications of grazing by native herbivores and livestock for conservation of grassland biota. Austral Ecology, 41: 455– 464. doi: 10.1111/aec.12337

Jones, D. L. (2006) New taxa of Australian Orchidaceae. Australian Orchid Research 5:75-76.

Keely, C. C., Hale, J. M., Heard, G. W., Parris, K. M., Sumner, J., Hamer, A. J. and Melville, J. (2015) Genetic structure and diversity of the endangered growling grass frog in a rapidly urbanizing region. Royal Society Open Science 2.

Kirkpatrick, S., Gelatt, C. D., and Vecchi, M. P. (1983) Optimization by Simulated Annealing. Science 220:671-680.

Kutt, A. S., Dalton, K. and Wills, T. J. (2016) Identification of reliable predictors of golden sun moth Synemon plana habitat over multiple survey years can benefit conservation, restoration and surveys for new populations. Journal of Insect Conservation 20:691-699.

Li, Y., Lancaster, M.L., Cooper, S.J.B., Taylor, A.C. and Carthew, S.M. (2015) Population structure and gene flow in the endangered southern brown bandicoot (*Isoodon obesulus obesulus*) across a fragmented landscape. Conservation Genetics 16, 331–345.

Lobert, B. 1985. The Ecology of the Southern Bandicoot in South-eastern Australian Heathland. Monash University, Clayton, Victoria.

Ludwig, D., Mangel, M. and Haddad, B., (2001). Ecology, conservation, and public policy. Annual Review of Ecology and Systematics, 32(1), 481 - 517.

MacHunter, J., Regan, T.J., Hauser, C. and Sinclair, S. (2019). Structured Decision Making for guiding management of MSA species and communities. Unpublished Client Report for the MSA Program. Arthur Rylah Institute for Environmental Research, Department of Environment, Land, Water and Planning, Heidelberg, Victoria.

Martin, T.G., Burgman, M.A., Fidler, F., Kuhnert, P.M., Low-choy, S., McBride, M. and Mengersen, K. (2012) Eliciting expert knowledge in conservation science. Conservation Biology, 26(1), 29 - 38.

McBride, M.F. and Burgman, M.A. (2012) What Is expert knowledge, how is such knowledge gathered, and how do we use it to address questions in landscape ecology? Expert Knowledge and Its Application in Landscape Ecology (eds A.H. Perera, C.A. Drew & C.J. Johnson), pp. 11-38. Springer New York.

McCarthy, M. A., and Thompson, C. (2001) Expected minimum population size as a measure of threat. Animal Conservation 4:351-355.

Menkhorst, P. (1990) Distribution and conservation status of bandicoots in Victoria. Pages 51-50 in J. H. Seebeck, P. R. Brown, R. L. Wallis, and C. M. Kemper, editors. Bandicoots and Bilbies. Surrey Beatty & Sons Pty Ltd, Chipping Norton.

Morgan, J. W. (1995a) Ecological-studies of the endangered *Rutidosis leptorrhynchoides*. 1. Seed production, soil seed bank dynamics, population-density and their effects on recruitment. Australian Journal of Botany 43:1-11.

Morgan, J. W. (1995b) Ecological-studies of the endangered *Rutidosis leptorrhynchoides*. 2. Patterns of seedling emergence and survival in a native grassland. Australian Journal of Botany 43:13-24.

Morgan, J. W. (1997) The effect of grassland gap size on establishment, growth and flowering of the endangered *Rutidosis leptorrhynchoides* (Asteraceae). Journal of Applied Ecology 34:566-576.

Morgan, J. W. (1999a) Effects of population size on seed production and germinability in an endangered, fragmented grassland plant. Conservation Biology 13:266-273.

Morgan, J. W. (1999b) Have tubestock plantings successfully established populations of rare grassland species into reintroduction sites in western Victoria? Biological Conservation 89:235-243.

Mueck, S. G. (2000) Translocation of Plains Rice-flower (*Pimelea spinescens ssp. spinescens*), Laverton, Victoria. Ecological Management and Restoration 1:111-116.

O'Hagan, A., Buck, C.E., Daneshkhah, A., Eiser, J.R., Garthwaite, P.H., Jenkinson, D.J., Oakley, J.E. and Rakow, T., (2006) Uncertain judgements: eliciting experts' probabilities. John Wiley & Sons.

O'Shea, M. B. (2013) Evaluating the effectiveness of slavage and translocation of Striped Legless Lizards. Depaerment of Sustaiability and Environment, Heidelberg, Victoria.

O'Shea, M. B. (2016) Population viability analyses for urban Striped Legless Lizard populations in the western Melbourne managment cluster. Victoria University.

Possingham, H. P., and Gepp, B. (1993) The application of Population Viability Analysis to assess managment options for the southern brown bandicoot (*Isoodon obesulus*) in the south-east of South Australia. Pages 633-638 in International Congress on modelling and simulation. UniPrint, Perth, Western Australia.

Reynolds, D. M. (2013) Factors affecting recruitment in populations of Spiny Rice-flower (*Pimelea spinescens Rye subspecies spinescens*) in Victoria's natural temperate grasslands: relationships with management practices, biological and ecological characteristics. Victoria University.

Reynolds, D. M, Walsh, N ., Mueck, S., Leahy, J. and Craigie, V. (2017) Burning *Pimelea spinescens* (V1_05.17). Trust for Nature. Melbourne, Victoria.

Richter, A., Osborne, W., Hnatiuk, S. and Rowell, A. (2013) Moths in fragments: insights into the biology and ecology of the Australian endangered golden sun moth Synemon plana (*Lepidoptera: Castniidae*) in natural temperate and exotic grassland remnants. Journal of Insect Conservation 17:1093-1104.

Rose, L. E., Heard, G. W., Chee, Y. E. and Wintle, B. A. (2016) Cost-effective conservation of an endangered frog under uncertainty. Conservation Biology 30:350-361.

Scarlett, N. H., and Parsons, R. F. (1990a) Conservation biology of the southern Australian daisy *Rutidosis leptorrhynchoides*. Pages 195-205 in T. W. Clark and J. H. Seebeck, editors. Management and Conservation of Small Populations. Chicago Zoological Society, Brookfield.

Unofficial

Scarlett, N. H., and Parsons, R. F. (1990b) Conservation biology of the Southern Australian Daisy *Rutidosis leptorrhynchoides.* Pages 195-205 in T. Clark and J. Seebeck, editors. Management and conservation of small populations: Proceedings of a conference, Melbourne, Australia, September 26 and 27, 1989. Chicago Zoological Society, Brookfield, Illinois.

Schultz, N., Keatley, M., Antos, M., Wong, N., Moxham, C., Farmilo, B. and Morgan, J. W. (2018) The golf ball method for rapid assessment of grassland structure. Ecological Management & Restoration 18:134-140.

Scroggie, M., Peterson, G. N. L., Rohr, D. H., Nicholson, E. and Heard, G. W. (2019) Disturbance has benefits as well as costs for fragmented populations of a cryptic grassland reptile. Landscape Ecology 34:1949-1965

Sinclair, S. J. (2010) National Recovery Plan for the Large-fruit Groundsel *Senecio macrocarpus.* Department of Sustainability and Environment, Melbourne, Victoria.

Southwell, D. M., Lechner, A. M., Coates, T. and Wintle, B. A. (2008) The sensitivity of population viability analysis to uncertainty about habitat requirements: Implications for the management of the endangered Southern Brown Bandicoot. Conservation Biology 22:1045-1054.

Walsh, N. G. (1999) Senecio. In: Walsh, N. G. and Entwisle, T. J. editors. Flora of Victoria vol 4: Dicotyledons Cornaceae to Asteraceae. Inkata Press, Melbourne.

Walters, C. J. (1986) Adaptive management of renewable resources. Macmillan Publishers Ltd., New York.

Young, A. G., Brown, A. H. D. and Zich, F. A. (1999) Genetic structure of fragmented populations of the endangered daisy *Rutidosis leptorrhynchoides.* Conservation Biology 13:256-265.

Young, A. G., and Murray, B. G. (2000) Genetic bottlenecks and dysgenic gene flow into re-established populations of the grassland daisy, *Rutidosis leptorrhynchoides*. Australian Journal of Botany 48:409-416.

Young, A. G., and Pickup, M. (2010) Low S-allele numbers limit mate availability, reduce seed set and skew fitness in small populations of a self-incompatible plant. Journal of Applied Ecology 47:541-548.

Zamin, T. J., Jolly, A., Sinclair, S., Morgan, J. W. and Moore, J. L. (2018) Enhancing plant diversity in a novel grassland using seed addition. Journal of Applied Ecology 55:215-224.

www.delwp.vic.gov.au

www.ari.vic.gov.au