

Eastern Barred Bandicoot PVA

Workshop, Zoos Victoria, Sept. 30 – Oct. 2

2012



This document reports on a workshop hosted by Zoos Victoria from September 30 to October 2, 2012. The workshop, which brought together members of the Eastern Barred Bandicoot Recovery Group, captive management specialists and scientists experienced in bandicoot biology, explored a range of recovery strategies for the species using the computerised population simulation program *Vortex* (Lacy *et al*, 2009). Staff of the IUCN/SSC/Conservation Breeding Specialist Group facilitated the workshop and constructed and analysed the models. The results of these analyses are to be made available to the recovery group as a planning tool.



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A contribution of the IUCN/SSC Conservation Breeding Specialist Group and Zoos Victoria, in collaboration with the Eastern Barred Bandicoot Recovery Team and the University of Melbourne.

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Executive Summary

Introduction

Mainland Eastern Barred Bandicoots (*Perameles gunnii*) are considered extinct in the wild and have been the subject of recovery action in Victoria for more than two decades. This work has been well documented and reviewed elsewhere (see Backhouse, 1992; Winnard & Coulson, 2008; DSE, 2009). The principle focus of recovery has been the re-stocking and management of a number of wild sites.

Zoos Victoria hosts and coordinates the captive program for mainland Eastern Barred Bandicoots (EBBs), which provides animals for release as part of this Department of Sustainability and Environment (DSE) Recovery Program for the species in Victoria. Forward planning for the captive program requires knowledge of the likely direction and needs of the release program into the future.

The following pages describe the outcomes of a Population Viability Analysis (PVA) workshop for mainland Eastern Barred Bandicoots, hosted by Zoos Victoria from September 30 - October 2, 2012. The purpose of the exercise was to explore, using computerised simulation models, what it might take to establish a population of mainland EBBs that could remain viable over the long-term, independent of the captive population, and what the captive population might need to provide to enable its establishment.

The workshop brought together members of the Eastern Barred Bandicoot Recovery Team, captive management specialists and scientists experienced in bandicoot biology. It was noted at the outset that key members of the Recovery Team were absent, but that their advice and input would be sought at a later date. The results of these analyses are to be made available to the Recovery Team to inform future planning discussions, which in turn will assist Zoos Victoria with its own plans.

Staff of the IUCN/SSC/Conservation Breeding Specialist Group were responsible for workshop design and facilitation, and for constructing, analysing and interpreting the models.

The modelling process proceeded through the following steps:

- defining the challenge;
- building relevant baseline models;
- identifying management scenarios for testing;
- running and interpreting the tests.

Each step is described in full in this document. The main results and conclusions are summarised below.

Defining the challenge

Since the decline of the species in Victoria, reintroduction has been attempted at eight separate sites, of which only two are known still to host EBBs. Environmental, demographic and genetic risk factors are likely to have been at play in the observed and repeated site extinctions. These were

discussed at length by the group with particular emphasis on the likely frequency and impact of environmental catastrophes such as fire and fox incursion.

Before identifying specific scenarios for modelling, a vision was developed by participants to help build consensus on the scope and direction of modelling and to encourage creative management alternatives for testing. It was agreed that for the vision to have practical value in this exercise the starting assumption should be that foxes will remain present in Victoria over the period considered.

50 Year VISION

In the absence of fox eradication, mainland Eastern Barred Bandicoots are genetically viable, in multiple population strongholds, removed from the threatened list and needing only limited management. The species occupies grassland and grassy woodlands in natural and modified ecosystems, on public and private land, not necessarily limited to its historic range. The bandicoot is recognised as an iconic species, promoting community pride and the conservation of grassy ecosystems in south-eastern Australia.

It was agreed that in order to envisage a meta-population able to overcome these challenges and deliver or contribute to the vision, it would be helpful to derive some insight into the following questions:

- how viable are populations likely to be at current and planned release sites, given the prevailing conditions?
- to what extent can risks to individual sites be moderated through regular supplementation events?
- with these things in mind, how big might a mainland population or meta-population of EBBs need to be?
- how should that meta-population be structured - few large or several small populations?
- how quickly would the meta-population need to grow to its target size?
- at what point might the captive population be no longer needed?

Building relevant baseline models

A generic baseline model was built for wild EBBs using data provided by workshop participants, a hypothetical founder base of 60, and a hypothetical carrying capacity of 1000, the latter to ensure that biological potential is not masked by the impact of limited carrying capacity. It excludes environmental catastrophes. This baseline allows us to explore species potential in the absence of extreme environmental events and to identify which life-history characteristics most influence

population performance. The latter can be helpful in priority setting for further data collection and in identifying targets for management intervention.

Models were also built for each current and planned EBB release site (Hamilton, Woodlands, Mount Rothwell, Werribee Open Range Zoo (WORZ)), and for captive populations at Melbourne Zoo and at WORZ. Catastrophes – i.e. periodic events expected to have an extreme impact on reproduction or survival are included in these. In addition, models were built and tested for three types of hypothetical site, with carrying capacities of 500, 800 and 4000, each complete with (hypothetical) site-specific catastrophes. Finally, a “source population” was constructed which broadly emulates the gene diversity measured in the current captive population. All remaining EBBs are either in captivity, or in wild populations founded from captivity. The current captive population, though not expected to paint an accurate picture, is the best guide available at time of writing, to the likely founder composition of these remaining stocks. All starting populations in the model are drawn from this source.

These site-specific models were tested individually and also combined to test meta-population scenarios.

Identifying management scenarios for testing

Three meta-population management scenarios were identified for testing, representing possible approaches to realising or contributing to, the vision described:

- Scenario 1.** NOW – includes Hamilton, Woodlands, Rothwell and WORZ
- Scenario 2.** BIG SITES includes the above but with 1 or 2 French Island-like sites added (and with the impact of cats excluded)
- Scenario 3.** HYPOTHETICAL SITES – includes the NOW sites but with 1, 2, 3 or 4 additional hypothetical sites at either $K=500$ or $K=800$.

In all cases, populations that go extinct are re-founded with 60 animals from captivity.

Running and interpreting the tests: the main conclusions

Individual site viability

In general, the models paint a picture of EBB populations as volatile and fluctuating, with the capacity for rapid growth but also rapid and regular decline. Where site carrying capacities are low ($K < \text{around } 300$), these characteristic fluctuations, the result of genetic and demographic stochastic or chance events, can drive populations to extinction relatively often, even in the absence of extreme environmental effects such as fire or fox incursion [*note that due to the degree of parameter uncertainty in the models, and the difficulty of estimated carrying capacity, it might be wise to add a generous margin of error to the $K=300$ value for any practical application*].

At larger carrying capacities ($K \geq 300$), populations can withstand their intrinsic genetic and demographic stochasticity over the 50-year period modelled, and can achieve a low probability of extinction ($PE_{50} < 0.05$) provided that the frequency and severity of environmental catastrophes

remains low. **None of the current, planned or hypothetical wild site models conform to these requirements**, that is, all show unacceptably high extinction risks over the period modelled ($PE_{50} < 0.05$). Of the existing release sites, Mount Rothwell shows slightly higher levels of viability and Hamilton the lowest *[though note that recent revision upwards of Hamilton's carrying capacity to at 120-150 (Weeks pers comm) would change this prognosis and increase the viability of the Hamilton model. This should be factored into any subsequent revisions of the models].*

This adds support to the currently advocated meta-population approach, which aims to spread extinction risk across a number of sites that can be linked in terms of strategic animal exchange and are unlikely to be affected by the same catastrophic event simultaneously. Periodic supplementation of sites can prevent extinction events and improve stability of numbers, though in general such supplementation events need to be often (every 5 years or so) and of around 8-15 individuals. *[Note that the models make assumptions about post-release survival and breeding success that may need to be revisited in advance of any practical application].*

Meta-population viability

Of the meta-population strategies modelled, the one which performs least well is that which includes only the smaller sites (Hamilton, Woodlands, Rothwell and WORZ). This requires the highest number of re-founding events and results in particularly low values for gene diversity and particularly high levels of inbreeding (though active management could reduce the latter to some extent). Extinctions can be reversed through re-founding, and in the models this is all done from a relatively secure captive population. Participants recognised that in order to harvest for re-founding from wild populations without placing them at risk, it is important to understand both the carrying capacity of each wild site and also what proportion of it is currently occupied. Models indicate that the acceptable margin of error for this in the smaller sites is relatively slight and advice from participants is that the difficulty of measuring these parameters accurately is high; taking a precautionary approach might therefore preclude any exit strategy for the captive population as a source of release animals, in this meta-population scenario.

Adding French Island to the mix improves the situation considerably, but only if the impact of cats proves negligible or significantly less uniform than is currently described in the models. Adding instead four sites at $K=800$ performs as well and in some areas better than this, for less overall carrying capacity. Only the addition of two French Island-like sites exceeds this success, but it does so only slightly, requires more than double the carrying capacity to do so, and is arguably somewhat less effective at spreading risk. On the basis of the tests carried out then, a meta-population strategy that includes four sites of around $K=800$, in addition to the smaller sites already in play, provides the best all-round results. *[Note that the success of $K=800$ reflects a combination of size and lower catastrophe susceptibility. Similar results might be seen at smaller sizes where catastrophes are fewer and rarer. This could be explored later in relation to real sites. Also, there may be other recovery objectives (e.g. cost, community expectation) not considered here that would increase the value of other strategies; this would need to be discussed by a broader group].*

Gene diversity retention and inbreeding accumulation

Inbreeding accumulation in all of the meta-population strategies reached levels that would be expected to depress fitness in a wild population ($F=0.20 - 0.30$). Periodic translocation is expected to reduce inbreeding accumulation at some of the smaller sites. Meta-population-wide gains in allelic

diversity resulting from fragmentation and isolation were observed in at least one of the scenarios (NOW plus four sites at $K=800$) and optimising rates of movement so that these are preserved whilst inbreeding accumulation is reduced, might be a useful focus for future modelling. However, this alone may not be sufficient to alleviate fitness issues related to inbreeding and lowered gene diversity. Though not tested here, periodic addition of animals from Tasmania would be expected both to improve gene diversity and reduce inbreeding accumulation across a mainland meta-population, however any gains would be lost rapidly at current population sizes and might best be considered as part of an expansion initiative.

Implications for the captive program

The length of time over which the captive population is required for release and/or insurance will depend on the choice of meta-population strategy and the rate of identification and preparation of wild release sites.

If one or other of the smaller scenarios is selected (e.g. NOW, or NOW plus one smaller site) a precautionary approach would see the captive population as a permanent fixture within the meta-population, both for ongoing insurance and as a periodic source of animals for release as wild populations fail. If one or other of the larger scenarios are selected and fully realised (e.g. NOW plus 4 sites at $K=800$ or NOW plus 1-2 Big Sites), the rate of re-founding required should be able to be met in an ongoing way through harvest from well-populated wild sites. In these scenarios, the captive population could play a short-term release role whilst sites are identified and seeded, followed by an insurance role while they become established, and beyond that could be wound-down and its role(s) replaced by other wild populations. Exact time periods are difficult to assess without additional information. In the models, new sites are founded at 5-year intervals, which stretch the site founding phase out to two decades in some cases. However, the modelled captive population can cope with a more aggressive harvesting regime if required. Assuming that this can be replicated in practice, the length of time for which the captive population would be needed will be determined by the rate of identification and preparation of wild sites.

In all of the meta-population scenarios modelled there is a release-intensive initial phase followed by periods of years in which no releases are required. This poses a challenge for the captive program, which must be able to produce large numbers of “surplus” animals in some years and none in others, and must do so without risking demographic crashes due to reproductive senescence or genetic deterioration resulting from regular bottlenecks. A flexible population control method such as pouch management is likely to be required, ideally in combination with a meta-population founding strategy that allows for a shorter but more intensive period of wild population establishment, should this be achievable .

Target population size

The IUCN/SSC advocates a range-wide, species-level approach to conservation planning wherever possible. This is particularly important when setting target population sizes and would ideally be the approach taken here; that is, any minimum target size set for mainland EBBs should take into account estimates of size, trends and management for Tasmanian counterparts. In the short-term this may be difficult to arrange due to geographical and administrative separation and interim targets for the mainland component may need to be agreed in isolation.

Based on a PVA meta-analysis of population viability analyses, Traill *et al.* (2007) advocate minimum sizes of several thousands (95% CI=3577-5129) as a general rule of thumb. This is not dissimilar from the MVP for long-term genetic security suggested by Franklin (1980) of a genetically effective size of 500, which in wild populations is expected to require a census population of around 5000 (Frankham, 1995c) and relates to an estimated “mutation-drift balance”, at which populations are expected to accumulate as much new diversity through mutation as they lose through drift (chance processes). In the case of mainland EBBs, which are a component of a wider taxon, a smaller size can be appropriate, in the context of occasional gene-flow across the broader taxon, and a target size of about half of this (N=2500) has been suggested. Pending a range-wide, species-level review of this target at some future point, the modelling results, which incorporate demographic uncertainty, support the proposed interim target of 2500, though indicate that this may require a carrying capacity of 4000-8000 depending on circumstance.

Population growth towards target size is required urgently to slow further genetic erosion, which will proceed rapidly at current population sizes and given the species’ short generation time.

Refining the models

It should be noted that these general conclusions have been drawn based on the parameters and estimates agreed during the workshop. There was considerable uncertainty attached to some of the parameters and those that have the greatest influence on population performance are as follows (in order of importance):

- site-specific frequency and severity of catastrophes
- site-specific carrying capacities
- age-specific mortality
- average percentage of females breeding in each 3-month time-step
- litter size in each 3-month time-step
- inbreeding severity and, informing this, starting genetic make-up and purging history

Further clarity around these would be expected to improve the value of the models.

Summary

In summary, the inherent volatility of EBBs and the environment which they inhabit, argues for a meta-population strategy in which individual populations show some resilience to genetic and demographic stochasticity (i.e. site K is at least 300 and ideally larger) and where the expected frequency and impact of catastrophes are not prohibitively high. Risk of extinction is ideally spread across several populations which are unlikely to be affected by the same catastrophe simultaneously. The strategy tested in this exercise that performed particularly well for a relatively modest boost in capacity, was that of adding to currently occupied and planned sites (Hamilton, Woodlands, Rothwell and WORZ), four additional sites each at K=800. In general, meta-population carrying capacities of $K \geq 3,000$ performed well both genetically and demographically and this might usefully be considered in the context of setting a target size for mainland EBBs, though ideally this would be done as part of larger, range-wide, species-level planning exercise incorporating Tasmanian stocks. Immediate population expansion is required to slow genetic deterioration and careful and innovative captive program design will be critical to the success of the project.

The models developed for this analysis are stored at CBSG and can be mobilised and modified as required to assist with future management planning.

Logical next steps would involve the identification of new, real scenarios that can be represented in the models. With additional information relating to cost, site management difficulty and other relevant factors, scenarios can then be compared in a transparent way using a list of objectives agreed and if necessary weighted, by decision-makers. A simple example of this kind of analysis is provided for illustration.

Introduction

[Note that this report represents the work and views of those present at the workshop. It was noted at the outset that key collaborators were not present and that their input would need to be sought before finalising any outcomes or recommendations arising from this work].

Mainland Eastern Barred Bandicoots (*Perameles gunnii*) are considered extinct in the wild and have been the subject of recovery action in Victoria for more than two decades. This work has been well documented and reviewed elsewhere (see Backhouse, 1992; Winnard & Coulson, 2008; DSE, 2009). The principle focus of recovery has been the re-stocking and management of a number of wild sites. To date, the establishment and maintenance of the species at these sites has been largely unsuccessful, with repeated extinctions in some areas. The Recovery Team for the species has analysed these extinctions to identify likely key requirements for release sites, in preparation for renewed attempts to recover the taxon in Victoria.

Zoos Victoria hosts and coordinates the captive program for mainland Eastern Barred Bandicoots (EBBs), which provides animals for release as part of this Department of Sustainability and Environment (DSE) Recovery Program for the species in Victoria. Forward planning for the captive program requires knowledge of the likely direction and needs of the release program into the future.

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Each step is described in full in this document.

Scene-setting presentations were provided at the start of days 1 and 2 of the workshop. Amy Winnard gave a presentation on the history and performance of populations at each of the release sites trialled to date and this information and the ensuing discussions, are summarised below. The slide presentation itself is attached as **Appendix V**. Andrew Weeks presented the results of a genetic analysis of remaining mainland EBB stocks and some of the information provided, in particular the likely founder number, was incorporated into the models.

Status Review

Site specific characteristics with respect to size, carrying capacity and current occupation, were discussed at length. **Table 1** summarises the status of past and present release sites with respect to these parameters, and of the captive population. Further details can be found in (DSE, 2009).

Carrying capacity is difficult to estimate and is likely to shift over time. Typical densities are considered to be 0.7-1.00 bandicoot per hectare with a range of 0.45-5.25 and large seasonal fluctuation [in Tasmania the range is estimated to be lower, at 0.35 – 2.35 bandicoots per hectare (Hill et al. 2010)].

Though all past and present release sites are included in the table, only Hamilton, Woodlands, Mount Rothwell and WORZ are currently being considered as candidates for future releases due to the availability of predator-proof fencing, now considered a pre-requisite for success.

Other conditions likely to favour a successful release project were considered to be:

- releasing a sufficient number of bandicoots;
- ensuring habitat quality:
 - by releasing at the appropriate point in what tend to be cyclic climatic conditions;
 - by controlling kangaroos and rabbits to prevent over-grazing;
- stability of staff and management regime to ensure ongoing favourable conditions.

It was noted that bandicoots seem to be able to cope both with drought and some over-grazing, in the absence of foxes.

Table 1. Summary of site-specific characteristics with respect to size, estimated carrying capacity and current occupation.

LOCATION	AREA (ha)	CURRENT POP	CARRYING CAPACITY	NOTES
HAMILTON	100	60	50-90	90 is the maximum estimate recorded (recent data suggests 120-150 may be more likely – this needs to be factored into future models).
MT ROTHWELL	430	250 +/- 100	200-300	- Current population size is not known. - Carrying capacity is estimated and may be too low based on recent analyses for Hamilton.

LOCATION	AREA (ha)	CURRENT POP	CARRYING CAPACITY	NOTES
				<ul style="list-style-type: none"> - It is assumed that not all habitat is suitable. - Possible competition with other species.
WOODLANDS	300	0	250-350	<p>Past population at greater than 1/ha</p> <p>Release of 15-25 planned for mid-2013</p>
WORZ	100	0	90-120	<p>Releases planned for 2012 and 2013.</p> <p>Assumptions:</p> <ul style="list-style-type: none"> - considered a “wild” site - management intervention (irrigation etc) - habitat restoration at site <p>Note: research opportunity around environmental conditions.</p>
FRENCH ISLAND	8000 expected suitable habitat (22000 total island)	0 (19 animals released as trial)	4000-8000	<ul style="list-style-type: none"> - Large cat presence and no substantial cat management. - Approximately 1 in 4 animals released contracted toxoplasmosis = suspected cause of death - May need to include higher mortality (both predation and disease) due to cats. - K is higher here than in previous estimates. K may be higher as EBBs exploit a range of habitats. - Trial release of sterile animals underway.
Mooramong	200	0	N/A	Unsuccessful – no predator-proof fence.
Floating Islands	85	0	N/A	Unsuccessful – predator control difficulties.
Lake Goldsmith	150	0	N/A	Unsuccessful – due to drought but also no predator-proof fence.
Lanark	111	0	N/A	Unsuccessful – drought and predator control difficulties.
Cobra Killuc	500	0	N/A	Unsuccessful – extinct due to drought, over-grazing and no predator-proof fence.
Captive breeding pop Melbourne Zoo & Werribee	65 enclosures	20.39.1	25 pairs plus young	Core breeding group

LOCATION	AREA (ha)	CURRENT POP	CARRYING CAPACITY	NOTES
Open Range Zoo				
Total captive enclosures (if required)	114 pens Including 65 at MZ/WORZ (25 breeding pairs)	114	114	Includes all animals (pre and post reproductive, display, and breeding). This number includes Melbourne Zoo, WORZ, other zoos (eg Healesville Sanctuary, Kyabram Fauna Park) and additional managed enclosures of varying sizes (eg Serendip Sanctuary, Mt Rothwell, Mooramong).

Note: WORZ_RELEASE = wild

Defining the challenge

Supporting the captive component of the Eastern Barred Bandicoot program has significant resource implications for Zoos Victoria and its partners. Planning for this would benefit from a greater understanding of what it might take to establish a meta-population of EBBs in the wild that would be viable over the long-term and independent of a captive population. This in turn would assist in planning for a captive population that would be capable of helping to establish such a wild meta-population, in terms of program length, size and frequency of releases and so on.

The Vision

Before identifying specific scenarios for modelling, a vision was developed by participants to help build consensus on the scope and direction of modelling and to encourage creative management alternatives for testing. It was agreed that for the vision to have practical value in this exercise the starting assumption should be that foxes will remain present in Victoria over the period considered.

VISION

In the absence of fox eradication, mainland Eastern Barred Bandicoots are genetically viable, in multiple population strongholds, removed from the threatened list and needing only limited management. The species occupies grassland and grassy woodlands in natural and modified ecosystems, on public and private land, not necessarily limited to its historic range. The bandicoot is recognised as an iconic species, promoting community pride and the conservation of grassy ecosystems in south-eastern Australia.

**Note that the need to incorporate reference to climate change impacts was recognised. It was agreed that allowing for establishment outside historical range would cover this pending more a more thorough consideration by a wider group.*

Exploratory models

To date, establishing permanent populations at wild sites has been problematic due to the frequency and severity of “catastrophic” events such as fox incursion and drought. Given the relatively small carrying capacities of some of the sites involved, the impacts of these catastrophes are likely to have been exacerbated by demographic stochasticity – chance fluctuations in birth and

death rates, and in sex-ratio, which have an increasingly dramatic effect on population dynamics as populations become smaller.

It was agreed that in order to envisage a meta-population able to overcome these challenges and deliver or contribute to the vision, it would be helpful to derive some insight into the following questions:

- how viable are populations likely to be at current and planned release sites, given the prevailing conditions?
- to what extent might risks to individual sites be moderated through regular supplementation events?
- with these things in mind, how big might a mainland population or meta-population of EBBs need to be?
- how should that meta-population be structured - few large or several small populations?
- how quickly would the meta-population need to grow to its target size?
- at what point might the captive population be no longer needed?

Some of the models pertinent to these questions were able to be built and run during the workshop, and these helped to shape the development of a number of alternative meta-population scenarios. Others have been run since the workshop and are reported on here for the first time. On the basis of this new information participants may consider generating additional scenarios for testing.

French Island and cats

French Island is currently under consideration as a release site and has a potentially large carrying capacity compared to other identified release sites. However, it also has a large cat population which cannot be removed. Cats are expected to be an extra mortality factor for EBBs. Based on a trial release, this extra mortality may operate both through direct predation and also through the transmission of the protozoal parasite, *Toxoplasma gondii* from cats to EBBs. However, the sample of release animals was small and the likely severity and age-specificity of cat-mediated mortality remains uncertain. Models were constructed to test a range of potential impacts of cats, to assess likely resilience of EBBs in the presence of cats. Based on the limited data available, the following French Island mortality scenarios were tested:

Table 2. Hypothetical parameters used to test the potential impact of cats on French Island EBBs.

Age-class (months)	Baseline Mortality rate (%)	Additional cat-mediated mortality (%)		
		Low impact uniform	High impact uniform	High impact selective
0-3	47	5	10	5
3-6	26	10	20	20
6-9	15	5	10	5
21-24	25	5	10	5
30-33	30	5	10	10

[Note the following veterinary advice from M. Lynch, provided post-workshop: “At this stage of our understanding it would make sense to say all age classes are equally susceptible. As experimental evidence is obtained this could be refined The on ground situation would suggest that cat impact should be included in the model. About 10% of EBBs presenting to Hamilton Vet lab in the 1980s died from toxoplasmosis”]. This should be factored into a future review of the models.

Meta-population management scenarios for testing

This section describes a range of potential meta-population management scenarios for EBBs. In thinking about and creating these, participants were keen to explore the relationship between site size and site security. Large fenced sites have the advantage of larger population sizes but management of the fences and predator control is more difficult to sustain. Smaller sites are disadvantaged in terms of population size but easier to sustain in terms of fence and predator management. It was noted that though drought is a catastrophe that could potentially strike all populations simultaneously, sites would be expected to show a differentiated response due to their particular locations or circumstances.

The management scenarios agreed for testing were as follows:

Management Scenario 1. NOW

This scenario incorporates sites which either currently host EBBs or are expected to shortly. French Island is not included in this scenario as its future as a release location still carries some uncertainty.

In this scenario, each site either begins with its current estimated population or, if it is currently or becomes vacant, it receives a founding population according to the rules laid out under **Conditions applicable to all sites** (below).

Table 3. Parameters used in the meta-population scenario NOW.

Sites included:	Current N:	Carrying capacity:	Included from:
Hamilton	60	50-90 (n.b. underestimate)	T=0
Mt. Rothwell	250	200-300	T=0
Woodlands	0	250-350	T=1
Werribee	0	90-120	T=1
Captive Population	SOURCE ONLY	60 animals per year	T=0

Management Scenario 2. French Island-like Additions

Scenario 2 considers the impact of adding to the current plans one or two very large island sites. French Island is the only such site currently under serious consideration. French Island carries a large cat population, the impact of which on EBBs is uncertain but may include increased mortality due to both direct predation and to infection with *Toxoplasma*. [Following a trial release of 19 animals, approximately 1 in 4 were found to have been infected with *Toxoplasma* and this may have been the principal cause of death]. A likely range of impacts is considered separately (see **Table 2.**) but for the purpose of this scenario the impact of cats is ignored and the “French Island-like” scenarios are considered cat-free.

Scenario 2a. “NOW” plus “French Island”

Scenario 2b. “NOW” plus two “French Islands”

Scenario 3: Adding hypothetical sites of K=500-800

This scenario considers the impact on meta-population viability, of adding, at five year intervals, one to four additional site at carrying capacities of 500 and 800.

Scenario 3a: “NOW” plus 1, 2, 3.....N populations of K=500, added at 5-year intervals.

Scenario 3b: “NOW” plus 1, 2, 3.....N populations of K=800, added at 5-year intervals.

Conditions applicable to all scenarios

The scenarios explored involve the release of captive bandicoots to different combinations of wild sites. The sites considered vary in size and character – some are real and some hypothetical.

In all scenarios:

- The captive population is the sole source of release animals and as such integral to the meta-population. However, population viability (and therefore success/failure) is reported for the populations outside captivity only.
- When populations go completely extinct (i.e. no EBBs left in the habitat), re-founding begins at the beginning of the next winter.
- Re-stocking/introductions are universally of 60 animals, released in two batches of 30, one in winter and one in spring to ensure two good breeding seasons at the outset.
- In the model, release animals are sub-adult (3 months/one-time-step old) bandicoots with an even sex-ratio (i.e. 15.15 in each batch).
- A 20% mortality rate is assumed for all translocation events.
- The captive population has a finite capacity for generating release animals, which is estimated to be around 60 animals per year. In year where more than one population requires founding or re-founding, sites are prioritised on the basis of their importance to the overall meta-population. That is, sites with larger carrying capacities are ranked higher. The order of priority is as follows:

1. French Island
2. Hypothetical large sites (500-800) ha
3. Mt Rothwell
4. Woodlands
5. WORZ
6. Hamilton

Note that though not all sites are included in every scenario, the order of priority remains constant throughout.

Additional Notes

A range of other considerations were discussed but eventually discarded to reduce complexity. These are recorded here and may be revisited later.

Preventing over-harvesting

Seeding or re-seeding sites from other wild sites was discussed. It is important to ensure that any site is able to bear the harvest taken. In the models this might be approached by preventing harvest from any wild site which is at or below 70% of carrying capacity. Ensuring this in reality would be challenging due to the difficulty of estimating both standing population size and carrying capacity at any given point in time.

In this analysis, all founding and re-founding events involve only captive animals.

Response to population declines

In the case of, for example, a population dropping to 30% of carrying capacity, the group discussed two potential approaches:

- 1) Supplement to reverse the decline.
- 2) Remove remaining animals to a more secure site.

Again, though these rules can be applied in the models, applying them in the field is problematic. Also, responses may need to be site specific. For example, at a larger site, a substantial decline may indicate a site event or condition which requires attention, whereas at a smaller site it may result from demographic stochasticity – chance fluctuations in birth and death rates that would go unnoticed in a larger population.

After discussion it was agreed that the models would allow populations to decline to zero without intervention, but with a re-founding event three-years later. This strategy was considered to mirror previous and likely future situations on the ground, whereby it takes some time to establish that animals are no longer present and to review and fix the reasons for this. *[This time-lag presented too many modelling challenges, and therefore in the scenarios modelled here, re-founding took place the following (early) winter after the last EBB disappeared from the population].*

Managing gene flow to reduce inbreeding

Where the meta-population is designed to include a number of smaller sites (e.g. $K \leq 500$), regular exchanges of animals will be required to prevent the negative impact of inbreeding accumulation. Maintaining a restricted level of gene-flow throughout the meta-population would be expected to have a beneficial impact on the long-term survival and health of all units, and on the overall allelic diversity of the meta-population. At a later date, optimal rates that take into account both site-specific and inherent demographic challenges, could be explored using the models. Additional information on survival and reproductive performance of translocated animals would be useful for this.

Building the Models: Parameters and Characteristics

Vortex Simulation Model

Computer modelling is a valuable and versatile tool for quantitatively assessing risk of decline and extinction of wildlife populations, both free ranging and managed. Complex and interacting factors that influence population persistence and health can be explored, including natural and anthropogenic causes. Models can also be used to evaluate the effects of alternative management strategies to identify the most effective conservation actions for a population or species and to identify research needs. Such an evaluation of population persistence under current and varying conditions is commonly referred to as a population viability analysis (PVA).

The software used in these analyses is the simulation program *Vortex* (v9.99b) (Lacy et al., 2009). *Vortex* is a Monte Carlo simulation of the effects of deterministic forces as well as demographic, environmental, and genetic stochastic events, on small wild or captive populations. *Vortex* models population dynamics as discrete, sequential events that occur according to defined probabilities. The program begins by either creating individuals to form the starting population, or by importing individuals from a studbook database. It then steps through life cycle events (e.g., births, deaths, dispersal, catastrophic events), for each individual and typically on an annual basis. Events such as breeding success, litter size, sex at birth, and survival are determined based upon designated probabilities that incorporate both demographic stochasticity and annual environmental variation. Consequently, each run (iteration) of the model gives a different result. By running the model hundreds of times, it is possible to examine the probable outcome and range of possibilities. For a more detailed explanation of *Vortex* and its use in population viability analysis, see Lacy (1993, 2000) and Miller and Lacy (2005).

The following section describes in detail the parameters used to build a series of eastern barred bandicoot models in *Vortex*, for the purpose of exploring the questions described in the previous section.

Baseline Models for Eastern Barred Bandicoots

Two baseline models were built for these analyses:

Baseline 1: a model that describes a “generic” free-living bandicoot population, in the wild and in the absence of environmental catastrophes and predation by exotic predators.

Baseline 2: a model that describes a captive population, managed under current husbandry regimes and using standard approaches to the management of population numbers, inbreeding and gene diversity retention.

General Model Parameters

The following parameters are common to both baseline models:

<u>Number of iterations:</u>	500 (1000 for most of the meta-population scenarios)
<u>Number of years:</u>	50 years (200 time steps of approximately 3 months or 91 days)
<u>Extinction definition:</u>	Only one sex remains
<u>Number of populations:</u>	Single population
<u>Percent males at birth:</u>	50%(no evidence found to date that suggests sex ratio at birth differs statistically from 50:50).
<u>Catastrophes:</u>	Not included in baseline models
<u>Harvest:</u>	Not included in baseline models
<u>Supplementation:</u>	Not included in baseline models
<u>Definition of “1 year”:</u>	1 “Vortex Year” = 3 months or approximately 91 days. Vortex proceeds in annual time-steps. EBBs are a short-lived species, begin breeding before the age of 1 year and can breed several times each year. Annual time-steps may not provide the resolution needed. Following a previous PVA (Lacy & Clark, 1990), 3-month time-steps are used in the baseline models here and wherever required, model inputs have been modified to take account of this.

Wild Model

Initial population size (N_i): 60 sub-adult founders

In discussion it was agreed that new populations of eastern barred bandicoots would be seeded with 60 sub-adults, released in two batches of 15.15, across two consecutive seasons (winter and spring to maximise initial breeding success). Sixty was selected because this is the estimated available harvest from captivity.

Unless otherwise specified, *Vortex* assumes all individuals in the initial population to be founders. As far as is known, all remaining EBBs, whether in captivity or in the wild, originate from 19 captive population founders. Using the default setting in *Vortex* will therefore overestimate gene diversity and underestimate inbreeding in the modelled populations. In later analyses this is addressed by drawing founders from a hypothetical gene pool with reduced gene diversity. However, this level of complexity is not required in the initial baseline and the *Vortex* defaults are applied instead.

Carrying capacity (K): 1000

This was selected in the baseline to remove the otherwise distorting impact on population dynamics of constrained carrying capacity. A plausible range of carrying capacities is tested elsewhere.

Mating system: Polygyny (maximum of 5 female mates per male per 3-month period)

From Peter Courtney: reproductive parameters, number of female mates per male per three months depends on the home range size; males have been considered to have ranges of 13 hectares and females 2.4 so could have access to the ranges of 5 plus females.

Age of first offspring: 2 time-steps (i.e. 6 months) for both sexes

Both the recovery plan and the previous PVA cite age at first breeding as four-and-a-half months – in the middle of the second time-step. *Vortex* requires the average age of first reproduction, not the age of sexual maturity or the earliest reproductive age observed and has been set slightly later – at the start of the third time-step.

Percent adult females breeding in a 3 month period: 90% at low density dropping to 80% at $N=K$ in winter and spring; 50% at low density dropping to 44% at $N=K$ in summer and autumn; $EV=10\%$

The percentage of adult females breeding in any three-month period is expected to vary both with population density and according to season. Differences between sites, small sample sizes and lack of firm estimates for standing population sizes and carrying capacity, make it difficult to estimate the boundaries and shape of these effects. Participants agreed on the following parameters based on the data available (provided by A. Winnard).

- **Summer and autumn:** percent breeding ranges from 50% at low density to 44% at high density.
- **Winter and spring:** percent breeding ranges from 90% at low density to 80% at high density.
- **Allee and steepness parameters** are set the same in both ($A=1$; $B=8$), describing an acute decrease in reproductive success both at very low and at very high densities. Though there were no data to support these specific parameters it was agreed that there were likely to be effects on reproductive output at both of these extremes.

Environmental variation causes substantial fluctuations in reproductive performance. However, participants considered that most of this effect is due to seasonal variation and to “catastrophes” such as drought. These are dealt with separately in the models and EV values are therefore set at a relatively modest 10%.

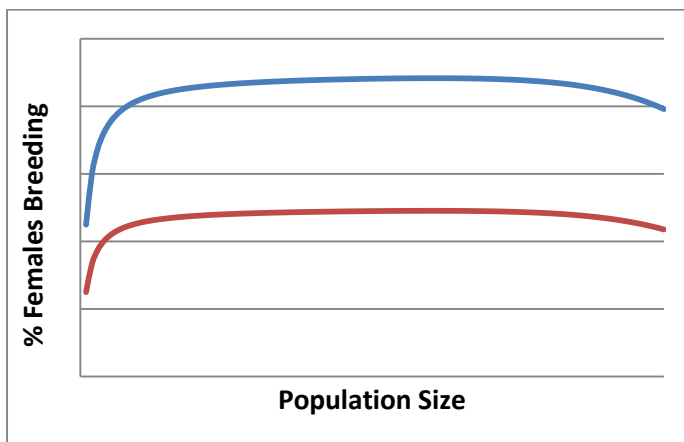


Figure 1 : Shape of the density dependent effect on female reproduction assumed in the wild models. The lower curve describes summer and autumn, the upper curve winter and spring.

$$= ((90 - ((90 - 80) * ((N/K)^8))) * (N/(1+N)) * ((Y\%4=3) + (Y\%4=0))) + ((50 - ((50 - 44) * ((N/K)^8))) * (N/(1+N)) * ((Y\%4=1) + (Y\%4=2)))$$

Where :

Y%4 = 0 is spring

Y%4 = 1 is summer

Y%4 = 2 is autumn

Y%4 = 3 is winter

Variation due to demographic stochasticity is built into the model.

Percent adult males in the breeding pool: 100%

In the absence of information to the contrary it is assumed that all adult males are available for and capable of, breeding.

Maximum number of litters per year: 4 (maximum of 1 in each 3-month time-step)

The national recovery plan reports up to 5 litters per year (citing Seebeck, 1979), whilst elsewhere it is suggested that under favourable conditions females may produce 3 or more, citing both Seebeck (1979) and Dufty (1994). Reproduction is described as year-round but with a depression in late summer and in years experiencing low rainfall or drought. Elsewhere (Lacy & Clark, 1990) interbirth interval is described as 70-90 days. At present we are defining a “year” as 3 months, with a specified chance of each female breeding in each “year”, leading to a maximum of 4 litters per year.

Number of progeny per litter: maximum 4; mean = 1.9 in summer and autumn; mean = 2.5 in winter and spring; EV=0.6

Hill *et al.* (2010) record litters of 1-5 with an average size of 2-3. The previous PVA report (Lacy & Clark, 1990) uses a mean litter size of 2.20 with sizes distributed as: 1 = 17%; 2 = 50%; 3 = 29%; 4 = 4%. Based on recent data collected by A. Winnard, across several years and in a growing population, model litter sizes are varied seasonally as follows:

- summer and autumn: mean = 1.9 (n=27)
- winter and spring: mean = 2.5 (n=79)

Additional variation due to environmental factors (EV) is set at 0.6, calculated from the data to exclude variation attributable to demographic stochasticity, which is automatically built into the model.

Mortality Parameters

Mortality rates: *Age specific* (same rates are used for both sexes)

Values used in a previous PVA (Lacy & Clark, 1990) were in part derived from real data and in part estimated. Both methods were based on observations and inferences relating to a population that had been declining at an annual rate of 25% and so the values used are not relevant here. Two sources of data were considered during the workshop: data collected by Amy Winnard as part of a

PhD project and data collected previously by Simone Jenkins as part of a MSc study, some of which was published in Todd *et al.*, 2002.

The data were combined to provide the following estimated mortality schedule:

- Juvenile (0-3 months - time-step 1): 47% (EV =15)
- Sub-adult (3-6 months – time-step 2): 26% (EV= 8)
- Adult (6 months – 2 yrs): 15% (EV=7)
- Adult (2 yrs – 2 yrs 9 months) = 25% (EV=7)
- Adult (2 yrs 9 months onwards) = 40% (EV=7)

The function used to describe mortality in the adult age-classes is as follows:

$$15+[(A>=8)*10]+[(A>=11)*15]$$

This schedule results in few animals surviving beyond the age of 3 years (4% in deterministic models). This accords with experience from the field (Winnard pers. comm.).

Variation due to demographic stochasticity is built into the model.

Inbreeding depression: Yes

Vortex models inbreeding as a reduction in juvenile survival. Ralls *et al.* 1988 analyzed studbook data for 38 captive mammal species and found the median lethal equivalents to be 3.14 LE. The effects of inbreeding are expected to be reduced in captivity as a result of the low-stress environment and supportive management. O’Grady *et al.* (2006) concluded that 12 lethal equivalents spread across survival and reproduction is a realistic estimate of inbreeding depression for wild populations.

A more optimistic approach is applied in the wild baseline (6.0 LEs applied to juvenile mortality), though this parameter is also tested as follows:

6.0 LEs applied to survivorship and the equivalent applied as the following multiplier to % females breeding - $E^{-I*0.03}$

Concordance between environmental variation in reproduction and survival: Yes

This is the default setting used for wild populations in the absence of species- or population-specific data. This means that environmental variation in reproduction and survival are directly linked, such that ‘good’ time-steps (3-month periods) for reproduction are also ‘good’ time-steps for survival; conversely, ‘bad’ periods for reproduction are linked to ‘bad’ periods for survival (worst case scenario for environmental variation). Models were also run with these two factors uncoupled.

Maximum age: Age-class 12 (3 years)

Individuals are removed from the model after they pass the maximum age. *Vortex* assumes that animals can reproduce throughout their adult life and does not model reproductive senescence unless this is specified by the user. No senescent period has been included in the wild baseline model.

Captive Model

Initial population size (N_i): 114 (at stable age-structure)

The baseline model aims to emulate a generic captive population to which alternative management strategies can be applied. The 114 starting individuals are drawn from a hypothetical source population whose genetic representation of the 19 individuals thought to have founded the current captive population, emulates that calculated through analysis of the known part of the studbook pedigree. As the pedigree is incomplete, this paints an inaccurate picture of current genetic composition, but one which will to some extent temper the otherwise overestimation of gene diversity and underestimation of inbreeding.

Carrying capacity (K): 150 (also tested at 100, 125, 175)

Current carrying capacity estimates vary and need to take into account spaces at Melbourne and Werribee Zoos, Healesville Sanctuary, Halls Gap, Kyabram, Serendip and Mooramong. Within Zoos Victoria there is currently capacity for around 25 breeding pairs, plus offspring. In the models, founding events are set to require 60 sub-adults for release and the captive population is expected to be able to accommodate one release each year of this size. *[In order to limit the harvest in the founding stages of the meta-population models, initial releases to empty sites are staggered as needed].*

Reproductive Parameters

Mating system: Polygyny (maximum of 2 female mates per male per 3-month period)

In an intensively managed captive population the number of mates per male should be able to be closely controlled and, to prevent over-representation of specific genetic lines would generally be maintained at close to 1 mate per male, per breeding event.

Age of first offspring: Time-step 3 (9 months) for both sexes

This parameter is designed to represent the average age of first reproduction, not the age of sexual maturity or the earliest reproductive age observed. For the wild population, age at first breeding has been set at 6 months based on field observations (see above).

For captive animals median age of breeding for females is 11 months (n=162) and for males 13 months (n=127) (Courtney, 2012) though animals are capable of breeding earlier. These would correspond to the third and fourth time-steps respectively. However, life-table data show a marked increase in offspring production at around 9 months for females and 11 months for males – both periods falling into time-step 3, and this value is currently applied in the baseline.

Density-dependent reproduction: No

It is assumed that, as carrying capacity approaches, managers will intervene with pre-agreed population restraint measures.

Percent adult females breeding: 40%, 50%, 60%, 70%, 80% and 90% (EV set at 5% of value)

At each time-step, this value is used to determine the probability of each female breeding. To estimate a plausible value for this it is necessary to know a) how many adult females breed, on average, during each 3-month period, and also b) how many females had the opportunity to breed during each period.

The studbook shows 162 female breeders to 429 non-breeders (approximately 37%). However, this does not take account of the number of females given an opportunity to breed. In the absence of additional data this parameter was tested at 40, 50, 60, 70, 80 and 90%, with an EV of 5% of the value (this is a low EV to reflect the relative constant nature of the captive environment).

Variation due to demographic stochasticity is built into the model.

Percent adult males in the breeding pool: 100%

In the absence of information to the contrary it is assumed that all adult males are available for and capable of, breeding.

Maximum number of litters per year: 2

In the wild, 3-5 litters per year are suggested (see above). In captivity, 6 litters comprising a total of 12 young have been observed (P. Courtney, pers. comm.) However, in captivity, the number of litters produced each year is often under close management and it is this that we aim to emulate in the models. The draft captive management plan suggests a maximum of two litters per year, to provide for release without creating an over-supply.

Number of progeny per litter: Max = 4; Mean = 1.75; S.D. 0.68.

Studbook records (Courtney, 2012) show the following statistics for litter size:

Mean litter size (n=613) is 1.75 (S.D. 0.68), distributed as follows:

1 = 38%

2 = 50%

3 = 11%

4 = 1%

For incorporation into the meta-population model, the mean and standard deviation are required. Variation due to demographic stochasticity is built into the model.

Mortality Parameters

Mortality rates: Age specific (same rates are used for both sexes) EV = 5% of value

Life-tables calculated from studbook data (Courtney, 2012) provide an approximation of age-specific mortality rates for males and females in captivity. EBBs live considerably longer in captivity than in the wild with roughly 50% of individuals still alive beyond reproductive senescence. For simplicity, animals are culled in the models once they reach maximum breeding age though it should be noted that in the absence of an alternative strategy, these animals, which may be in considerable numbers, will need to be accommodated within captive carrying capacity.

The following mortality values are applied in the model.

- Juvenile (0-3 months yr): 10.0%
- Sub-adult (3-6 months): 5.0%
- Sub-adult (6-9 months): 5.0%
- Adult (9-36 months): 35%

EV is set at 5% of the mortality value to reflect relatively constant environmental conditions in captivity. Variation due to demographic stochasticity is built into the model.

Inbreeding depression: Yes

Vortex models inbreeding as a reduction in juvenile survival. The default value in Vortex is 3.14 lethal equivalents, of which 50% are assigned to lethal alleles and subject to purging. This value is the median LE calculated from studbook data for 38 captive mammal species (Ralls *et al.* 1988). In the absence of other estimates, these defaults have been applied in the captive model.

Concordance between environmental variation in reproduction and survival: No

For the captive model, reproduction and survival have not been directly linked – that is, that ‘good’ years for reproduction are not necessarily also ‘good’ years for survival.

Maximum age: Time-step 12 (3 years) for both sexes

Individuals are removed from the model after they pass the maximum age. *Vortex* assumes that animals can reproduce throughout their adult life and does not model reproductive senescence unless this is specified by the user. For captive populations, senescence begins at age around time-step 12 (3 years for both sexes). In captivity this may be followed by a period of senescence lasting up to age 5 or 6 years (time-steps 20-24). During this time animals are taking up physical space, but are not contributing further to the next generation. The presence of these senescent animals is not modelled in the baseline but they can be included in subsequent models.

Note that according to studbook records, at stable age-structure, and given past performance of the captive population, senescent animals might be expected to make up 50% of the standing population.

Genetic Management: management to K; static mean kinship; maximum allowable inbreeding $F=0.375$

Vortex models allow the incorporation of standard captive management approaches. A widely used strategy for maximising gene diversity in captivity is to use mean kinship values to prioritise individuals for breeding who are expected to contain rarer alleles. This has been applied in the baseline. Efforts are usually also made to restrain the rate of inbreeding accumulation by avoiding breeding between individuals whose offspring would carry a high inbreeding coefficient. In a closed population over time, inbreeding is inevitable. In these circumstances, setting the inbreeding threshold too low will constrain the ability of *Vortex* to find suitable pairs as the simulation progresses. The threshold here is set to $F=0.375$, which should be sufficient to stave off extreme inbreeding effects whilst allowing breeding to continue. And finally, the production of surplus

animals in captivity is most often achieved by calculating the breeding rate required to maintain the population at capacity, and setting up, each year, only the number of pairs required to achieve that rate. There are other methods of population control but management to capacity (K) is the one applied in the baseline at present.

Wild Baseline Model Performance

Deterministic output

The demographic rates (reproduction and mortality) included in the baseline model can be used to calculate deterministic characteristics of the model population. These characteristics reflect the biology of the modelled population in the absence of: stochastic fluctuations in demographic rates and environmental impacts; inbreeding depression; limitation of mates; and any immigration or dispersal. It is valuable to examine deterministic characteristics (lambda, generation length, and age structure) to assess whether they appear realistic for the species being modelled. Catastrophes are not included in this initial analysis as although all populations are expected to experience them, no two populations are thought to experience them with the same frequency or impact.

Field data indicate high female breeding rates and larger litter sizes in winter and spring, and lower female breeding rates with reduced litter sizes, in summer and autumn (Winnard, pers.com). Field data on density effects are scant, but participants predicted a small depression in reproductive rates at high densities. Fecundity (both percent of females breeding and mean litter size) are therefore varied both seasonally and with density, in the baseline model.

Model behaviour is consistent with this pattern, exhibiting high growth during the peak seasons (approximately 20% per 3-month period during winter and spring) and little or no growth during the low seasons (0.1% per 3-month period during summer and autumn). Table 4 summarises the growth measures assessed:

Table 4. Deterministic growth in the baseline (wild) model:

Growth Measure	Winter/Spring	Summer/Autumn
Ro (growth per generation)	2.39	1.01
T (generation time in years)	1 year 9 months	1 year 9 months
λ (lambda – annual growth rate)	1.20 (per 3-month time-step) (annual $\lambda = 2.108$)	1.00 (per 3-month time-step) (annual $\lambda = 1.004$)
r (instantaneous growth rate)	0.19 (per time-step)	0.00 (per time-step)

Observed wild growth rates of lambda = 1.04, 1.11, 1.16 and 1.29 per time-step, were calculated from field observations reported in the literature (see Appendix III). The model results accord reasonably well with this.

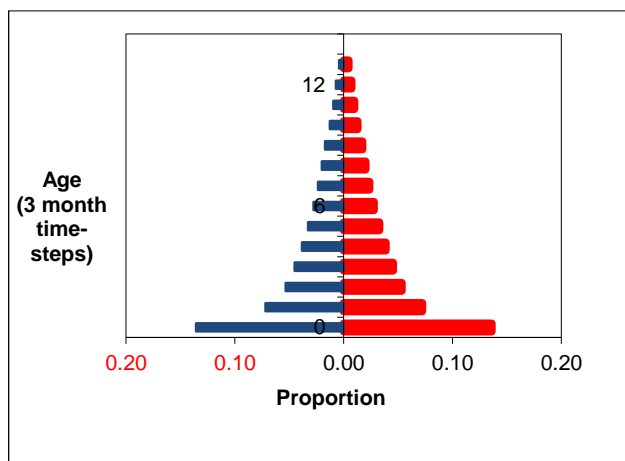


Figure 2 : Population age structure generated by the baseline model
Note that “age” is displayed in 3-month time-steps, i.e. “12” = 3 years.

In the modelled population, adults make up approximately 60% of the standing population (see Figure 2), with fewer than 10% of individuals born surviving beyond 9 time-steps or 2 years 3 months of age. This seems consistent with field observations (A. Winnard, pers.comm.) though again, few data are available for direct comparison.

Stochastic output

With stochastic fluctuations in both demographic and environmental influences, and inbreeding included at default levels, the modelled population continues to perform well, growing from an initial population size of 60 to the carrying capacity of 1000, over a period of 5-10 years. Mean growth rate over the 50-year time-frame is approximately 5% per time-step or 3-month period (stoc r =0.054). Population size tends to vary considerably over the 50-year time-frame (mean N =907; SD =144.73), however extinction risk is relatively low (PE_{50} =0.04). Gene diversity declines over the period from an initial 99.16% (that expected to be captured in a founder base of 60 individuals) to 89.97%. Figures 4 and 5 illustrate the average behaviour of the population over 50 years with respect to population size and gene diversity retention.

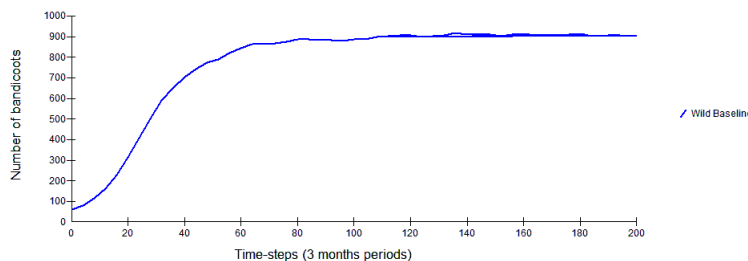


Fig 4. Mean population size over 50 years (200 time-steps) across 500 simulations of the Wild Baseline model.

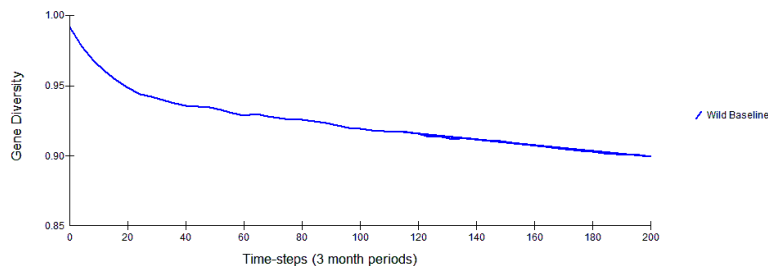


Fig 5. Mean gene diversity over 50 years (200 time-steps) across 500 simulations of the Wild Baseline model.

Note that the baseline emulates an **optimistic trajectory** for any eastern barred bandicoot population:

- a) it has a larger carrying capacity than most current or planned sites;
- b) it does not include any catastrophic events (such as a fire or fence breach);
- c) it starts with a founder base of 60 individuals (though it is likely that all available founders for future populations will derive from a recent bottleneck of approximately 15-19 founders).

The impact of these additional stresses is explored in detail in later sections, using this baseline model as a starting platform.

Sensitivity Testing

Wherever possible, real data have been used to inform model parameters. However, many of the parameters are estimates and subject to varying levels of uncertainty.

We are unable to reduce parameter uncertainty in this exercise as that would require further research and data collection. However, it is useful to know which of the areas of uncertainty has the greatest impact on model performance, so that we can:

- a) Understand where key pressure points in the species' biology and environment are.
- b) Design management interventions accordingly.
- c) Prioritise future research and data collection to improve the predictive value of the models.

To develop an understanding of where the key “pressure points” are, we can test the sensitivity of the models to each parameter in turn, by varying each across a plausible range of values.

The following sensitivity tests were carried out on the Wild Baseline model (BOLD indicates the baseline value):

Table 5: Values for Sensitivity Testing (BOLD indicates the Wild Baseline value)

Model Parameter	Range of Values Tested
Concordance between survival and reproduction	YES and NO
Inbreeding severity (number of lethal equivalents)	3.14 (captive default), 6.00 , 9.00, 12.00
% females breeding each year	50, 60, 70, 80, 90, 95*
Minimum age for females breeding	2 , 3, 4
Mean Litter Size	1.9, 2.2 (captive mean), 2.5**
Density dependence effects: Allee	0, 1 , 2, 3
Density dependence effects: reproductive decline approaching K	2, 4, 8 , 16
Carrying capacity (K)	50, 100, 150, 200, 300, 400, 500, 800, 1000 , 2000, 4000

*baseline value is a function involving density dependence and seasonal variation

**baseline value varies seasonally between 1.9 and 2.5

Results

The parameters exerting the most influence on population performance across the range of values considered, are carrying capacity, the percentage of females breeding and litter size. Inbreeding severity has a lesser but still potentially important effect. Uncoupling concordance between reproduction and survival has little impact. Likewise, changing the severity of impact of density dependence has little impact either at high or low densities, for the range of values considered.

The relative impact of these parameters on population growth rate is illustrated in **Figure 6**. The impact of carrying capacity is not included here but is dealt with separately below.

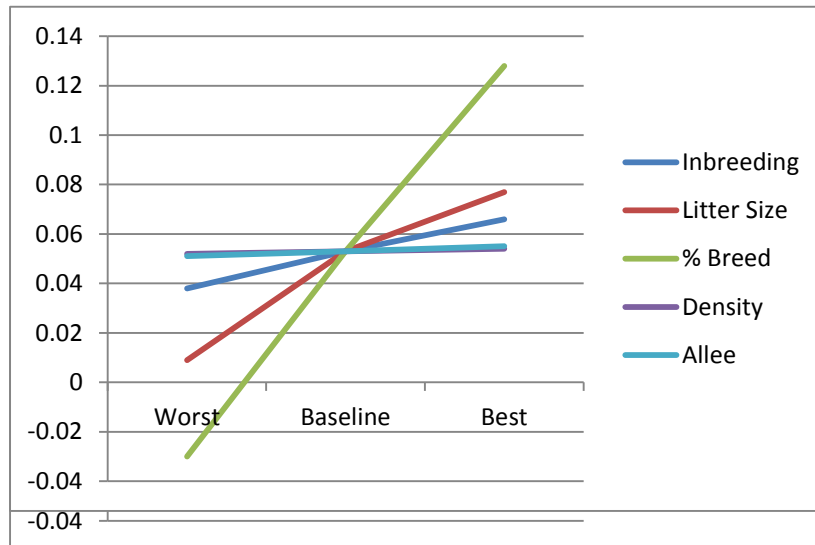


Fig 6. An illustration of the relative impact of different biological parameters on model growth rate. “Worst” and “Best” indicate growth rate at the upper and lower limits of the estimated range of plausible values for each parameter (where growth is measured as average r per 3-month time-step).

As carrying capacity is one of few characteristics able to be influenced directly via site selection and/or management, this is considered in more detail here.

Carrying capacity impacts

Reducing population size by limiting carrying capacity is expected to a) increase the rate of inbreeding and b) increase the influence of demographic stochasticity (chance fluctuations in birth and death rate and in sex-ratio) on population performance.

Carrying capacities of 50, 100, 150, 200, 300, 400, 500, 800, 2000 and 4,000 were trialled as part of the sensitivity tests reported here. The impact on mean population size over time is shown in **Figure 7** below, and a range of model parameters are compared across scenarios, in **Table 6**.

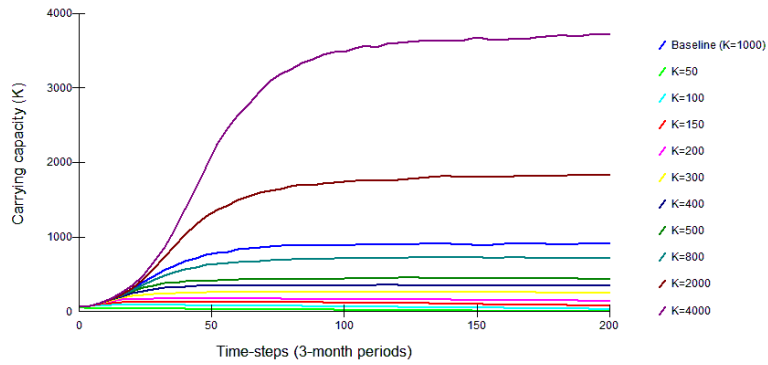


Fig 7. Mean population size over 50 years (200 time-steps) for a representative range of carrying capacities.

Table 6. Comparison of baseline model performance with varied carrying capacity.

Carrying capacity	Growth (stoc-r)	SD(r)	PE ₅₀	Mean N (Extant populations) at 50 years	Gene diversity at 50 years
K=50	-0.009	0.222	1.000	0	0
K=100	0.007	0.203	0.744	34.48	0.5280
K=150	0.021	0.194	0.284	78.07	0.6565
K=200	0.032	0.191	0.104	136.96	0.7425
K=300	0.041	0.189	0.048	248.32	0.8209
K=400	0.045	0.191	0.040	345.10	0.8464
K=500	0.048	0.191	0.044	429.65	0.8700
K=800	0.052	0.191	0.046	718.71	0.8907
K=1000 (baseline)	0.053	0.192	0.048	907.59	0.9007
K=2000	0.056	0.191	0.040	1834.08	0.9168
K=4000	0.041	0.175	0.034	3719.09	0.9265

In the modelled populations substantial gains are made in reducing probability of extinction and increasing average growth rate, by increasing carrying capacity. The effect is most pronounced below K=300 by which probability of extinction has fallen below 5%, a commonly applied threshold. Beyond this the gains continue but at a reduced rate. At K=1000 and above, gene diversity retention is above 90%, another standardly applied threshold. However, this is based on an initial founder base of 60 and therefore unlikely to be achievable in practice.

Table 7. Summary of sensitivity text results

Scenario	stoc-r	SD(r)	PE ₅₀	N-extant ₅₀	SD(Next ₅₀)	GeneDiv ₅₀
Baseline Values	0.053	0.192	0.048	907.59	153.40	0.9007
Wild_Inb_LOW	0.066	0.196	0.010	933.59	110.78	0.9010
Wild_Inb_HIGH	0.038	0.188	0.146	847.13	211.28	0.8972
Wild_Inb_MAX	0.025	0.187	0.294	809.92	237.64	0.9038
Wild_NoEV_Concord	0.049	0.168	0.022	919.04	152.19	0.8975

Scenario	stoc-r	SD(r)	PE ₅₀	N-extant ₅₀	SD(Next ₅₀)	GeneDiv ₅₀
Wild_DD_B2	0.051	0.189	0.034	886.13	176.89	0.8991
Wild_DD_B4	0.052	0.191	0.042	904.53	144.39	0.9024
Wild_DD_B16	0.054	0.191	0.042	922.56	123.86	0.9041
Wild_Allee_0	0.055	0.191	0.032	922.45	121.35	0.9017
Wild_Allee_2	0.053	0.191	0.052	895.26	169.50	0.8995
Wild_Allee_3	0.051	0.191	0.064	901.92	154.37	0.8953
Wild_Litter_1.9_0.65	0.009	0.177	0.414	624.37	325.27	0.8543
Wild_Litter_2.2_0.65	0.049	0.177	0.044	895.84	157.00	0.9022
Wild_Litter_2.5_0.65	0.077	0.185	0.008	941.43	96.71	0.9173
Wild_Fmin_repro_4	-0.038	0.209	0.974	167.38	197.16	0.7857
Wild_FBreed_50	-0.030	0.196	0.934	166.58	251.31	0.7960
Wild_FBreed_60	0.025	0.169	0.228	709.15	288.83	0.8791
Wild_FBreed_70	0.067	0.167	0.018	905.47	130.13	0.9144
Wild_FBreed_80	0.099	0.169	0	958.14	73.25	0.9263
Wild_FBreed_90	0.128	0.171	0	975.94	52.27	0.9313
Base_Wild.K(50)	-0.009	0.222	1	0	0	0
Base_Wild.K(100)	0.007	0.203	0.744	34.48	28.40	0.5280
Base_Wild.K(150)	0.021	0.194	0.284	78.07	45.38	0.6565
Base_Wild.K(200)	0.032	0.191	0.104	136.96	55.36	0.7425
Base_Wild.K(300)	0.041	0.189	0.048	248.32	64.27	0.8209
Base_Wild.K(400)	0.045	0.191	0.040	345.10	73.44	0.8464
Base_Wild.K(500)	0.048	0.191	0.044	429.65	88.76	0.8700
Base_Wild.K(800)	0.052	0.191	0.046	718.71	121.72	0.8907
Base_Wild.K(2000)	0.056	0.191	0.040	1834.08	287.20	0.9168
Base_Wild.K(4000)	0.041	0.175	0.034	3719.09	540.02	0.9265

Summary

Sensitivity analyses indicate that site carrying capacity, female breeding rates and litter size are key to population performance and greater certainty around estimates of these would be expected to enhance the predictive value of the models. Inbreeding severity is also important, though less so, and would also benefit from further study.

The size and shape of density dependence effects seems to carry less importance, though as so little is known about this from field studies the “plausible range” considered may have been too narrow.

Treatment of Founder Gene Pool

In the Wild Baseline model, the Vortex default application of genetic modelling is applied, whereby the starting population (60 sub-adults) consists of unrelated founders, each possessing two unique alleles, the fates of which are tracked through subsequent model simulations and used to calculate population gene diversity and individual inbreeding coefficients.

All current mainland eastern barred bandicoots derive from a small number of founders (estimated to be 19) once held in captivity and used to populate wild release sites. Throughout the course of EBB recovery management, animals have moved back and forth between wild sites and the captive population, but all derive from those 19 founder animals. As a result, the default assumption in *Vortex* – that all animals in the starting population are founders – will lead to an overestimate of gene diversity and an underestimate of inbreeding depression.

Records for captive eastern barred bandicoots are maintained in a studbook database, which allows relationships between all animals held in captivity to be tracked, including those subsequently released to the wild. This has the potential to provide valuable insights into the likely genetic make-up of populations at wild sites. Unfortunately the studbook pedigree is not sufficiently complete for this and alternative approaches have been sought. The following has been applied in all scenarios:

A hypothetical population of “animals” has been created, from which founders can be drawn to seed and to supplement wild sites and to establish the captive population.

Studbook analyses show 15 founders contributing to the current living population (rather than the 19 derived through other studies). In the absence of further information about the additional 4 individuals, only 15 are incorporated into the hypothetical source population here. This takes a precautionary approach and can be modified later should the additional information become available.

These 15 founders are assigned 2 alleles each, such that 30 alleles are incorporated into the hypothetical source population.

Studbook analyses indicate that those original 15 founders are no longer evenly represented in the captive population due to subsequent differential success amongst the lines. The 30 alleles are therefore assigned unequal frequencies in an attempt to reflect this. Given the incomplete pedigree, this assignment has been based on a number of assumptions:

- that the UNK portion of the pedigree has the same founder representations as the known portion;
- that the current proportions in the captive pop are representative of the proportions in the past when animals were released; and
- that the individuals chosen for release were representative of the entire captive population, and that there has been no genetic drift, selection or other changes in allelic frequencies in the wild since release.

While these assumptions are not realistic, without further information they are likely to more closely mirror the prospects of extant EBBs than the *Vortex* default assumption of unrelated founders to all populations.

A limitation of this approach is that although *Vortex* tracks both gene diversity and average inbreeding levels using the founders drawn from the customised source population, the model uses the default locus (treating all initial and supplemented individuals as unrelated founders) to apply inbreeding depression, thereby underestimating its impact in the models.

Using the hypothetical source population to initiate the Wild_Baseline model reduces gene diversity (by approximately 5% over the 50 year period) and increases inbreeding accumulation (from a population average of 0.0971 to 0.151 at the end of 50 years). A common default maximum threshold for captive populations is 0.125. Though in the model the elevated inbreeding level does not register as an impact on probability of extinction, population size or growth rate, we might expect it to have some impact on these factors at the levels projected. On the positive side, this population has experienced a severe bottleneck, which may have purged a proportion of the more deleterious alleles, thereby reducing susceptibility to inbreeding depression. This would benefit from further analysis.

Table 8. Comparison of performance for populations founded with 60 and with 15 founders.

Scenario	stoc-r	SD(r)	PE ₅₀	N-extant ₅₀	SD(Next) ₅₀	GenDiv ₅₀	F ₅₀
Base_Wild	0.053	0.192	0.048	907.59	144.13	0.9007	0.0971
Base_wild_ (from 15 fdrs)	0.053	0.190	0.036	901.78	151.32	0.8455	0.1515

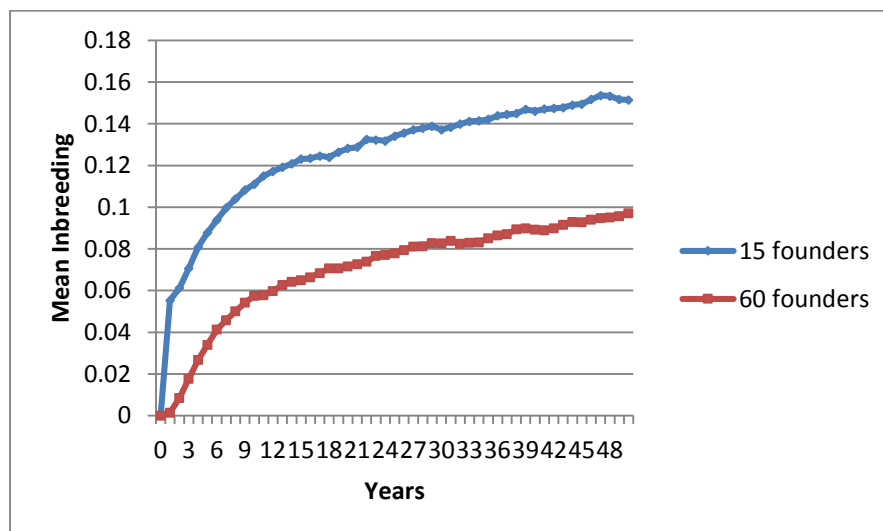


Fig 8. Comparison of inbreeding accumulation over time for populations initiated with 60 evenly represented and with 15 unevenly represented founders.

Treatment of Catastrophes

A range of influences were considered to be operating at current and potential release sites which could produce extreme mortality and/or reproductive rates in resident Eastern Barred Bandicoots. These are likely to have a considerable impact on population performance and are characterised as follows:

- Severe Fire.** Characterised as a burn over 90% of the reserve resulting in loss of animals through the direct impact of the fire and additionally as a result of fence breaches and consequent fox incursion. It was assumed that fences would be reconstructed after three months and foxes removed.
- Mild Fire.** Characterised as a burn over a smaller proportion of the reserve with no accompanying fence breach.
- Flood.** Characterised as a flood which knocks down a section of the fence allowing fox incursion. Fences are reconstructed quickly and foxes are removed after 3 weeks.
- Fox Incursion.** 1-2 foxes enter and are detected and removed swiftly.
- Fox Establishment.** Fox population becomes established, either through deliberate release or “natural colonisation (e.g. swimming to an in-shore island).
- Drought.** Characterised as a severe drought lasting for 3 years.

Other potentially catastrophic events were discussed but eventually discarded. It may be useful to revisit these in future, in the context of discussions about new sites. These were:

- **Flood:** the direct impact of inundation was discussed but for the sites under consideration this was agreed to be an unlikely cause of high mortality or reduced breeding.
- **Over-grazing or inappropriate land-management:** this was considered to be largely covered under the Environmental Variation provisions of the models, and likely only to impact as a secondary factor, for example in situations where other catastrophes such as fox incursion or drought are operating.
- **Contagious disease:** none were known and so incorporation into models was considered too difficult.
- **Heatwave:** extreme heat events over several days. Not included in the models at this stage but participants suggested contacting Mike Kearney for information on thermal tolerance.

Sites were considered to differ in the types of catastrophes to which they are exposed, and in the frequency and severity of those events. Site-specific catastrophe schedules are listed below:

Hamilton EBB Catastrophes

Catastrophe	Nature of impact	Frequency	Impact severity* – reproduction	Impact severity* - survival	Local/Global	Notes/queries
Severe Fire	Burn over 90% of reserve Fences compromised and foxes enter – fences reconstructed after 3 months and foxes removed.	Every 50 years	0-3 months – 0.50 3-6 months – 1.00	0.02 (i.e. 98% loss)	Local	Only occurs in summer:
Mild Fire	Burn over 33% of reserve (no fences compromised)	Every 15 years	0-3 months – 0.90 3-6 months – 1.00	0.80 (i.e. 20% loss)	Local	Occurs spring–autumn: - look up fire frequency
Fox Incursion	1-2 foxes enter reserve –detected within 3 days, removed within 2 weeks.	Every 5 years	No impact	0.90 (i.e. 10% loss)	Local	Assumes at least weekly checks and rapid response.
Drought	Severe drought occurring for 3 years.	Every 15 years	Summer: 0.1 Autumn: 0.1 Winter: 0.5 Spring: 0.5	0.70 (i.e. 30% loss)	Global	Climate change modelling required to update severity and frequency. Values may need adjustment. Re-look at data sets to determine drought impact (based on lowest breeding levels at Mooramong)

*(proportion of normal values)

Mount Rothwell

Catastrophe	Nature of impact	Frequency	Impact severity* – reproduction	Impact severity* - survival	Local/Global	Notes/queries
Severe Fire.	Burn over 90% of reserve. Fence breach and fox incursion – fences reconstructed after 3 weeks and foxes removed.	Every 40 years	0-3 months – 0.50 3-6 months – 1.00	0.20 (i.e. 80% loss)	Local	Only occurs in summer: - look up fire frequency for area
Mild Fire.	Burn over 33% of reserve (no fences compromised).	Every 15 years	0-3 months – 0.90 3-6 months – 1.00	0.80	Local	Occurs spring/autumn: - look up fire frequency
Flood.	Flood from neighbours property knocks down section of fence and foxes enter – fences reconstructed quickly and foxes removed after 3 weeks.	Every 5 years	No impact	0.90	Local	Get frequency records from Mt Rothwell.
Fox Incursion.	1-2 foxes enter reserve –detected within 1 day, fox removed within 1 week.	Every 10 years	No impact	0.95	Local	Assumes at least daily checks and rapid response.
Drought.	Severe drought occurring for 3 years.	Every 15 years	Summer: 0.10 Autumn: 0.10 Winter: 0.50 Spring: 0.50	0.90	Global	Climate change modelling required to update severity and frequency. Values may need adjustment. Look at data sets to determine drought impact (based on lowest breeding levels at Mooramong).

*(proportion of normal values)

Woodlands

Catastrophe	Nature of impact	Frequency	Impact severity* - reproduction	Impact severity* - survival	Local/Global	Notes/queries
Severe Fire.	Burn over 90% of reserve with associated fence breach. Fences reconstructed after 3 weeks and foxes removed.	Every 50 years.	0-3 months – 0.5 3-6 months – 1.00	0.20	Local	Only occurs in summer: - look up fire frequency for area
Mild Fire.	Burn over 33% of reserve (no fences compromised).	Every 15 years.	0-3 months – 0.90 3-6 months – 1.00	0.80	Local	Occurs spring/autumn: - look up fire frequency at Woodlands
Fox Incursion.	Fence cut and 1-2 foxes enter reserve –detected within 1 week, fox removed within 1 month.	Every year.	No impact	0.90	Local	Assumes at least weekly checks and a response.
Drought	Severe drought occurring for 3 years	Every 15 years	Summer: 0.10 Autumn: 0.10 Winter: 0.50 Spring: 0.50	0.70	Global	Climate change modelling required to update severity and frequency. Values may need adjustment. Look at data sets to determine drought impact (based on lowest breeding levels at Mooramong).

*(proportion of normal values)

French Island

Catastrophe	Nature of impact	Frequency	Impact severity* – reproduction	Impact severity* - survival	Local/Global	Notes/queries
Severe Fire.	Burn over 90% of habitat.	Every 50 years	0-3 months – 0.50 3-6 months – 1.0	0.20	Local	Only occurs in summer.
Mild Fire.	Burn over 10% of habitat.	Every 10 years	0-3 months – 0.90 3-6 months – 1.0	0.90	Local	Occurs spring/autumn.
Fox Establishment.	Fox population established on island through deliberate release and/or successful swimming.	Every 50 years	No impact	0.00 (i.e. 100% loss of population)	Local	Assumes no monitoring system for early detection.
Drought.	Severe drought occurring for 3 years.	Every 50 years	Summer: 0.10 Autumn: 0.10 Winter: 0.50 Spring: 0.50	0.90 (i.e. 10% loss of population)	Global	Climate change modelling required to update severity and frequency Values may need adjustment look at data sets to determine drought impact (based on lowest breeding levels at Mooramong)

*(proportion of normal values)

WORZ (Wild)

Catastrophe	Nature of impact	Frequency	Impact severity* – reproduction	Impact severity* - survival	Local/Global	Notes/queries
Severe Fire.	Burn over 90% of reserve.	Every 50 years.	0-3 months – 0.50 3-6 months – 1.0	0.20	Local	Only occurs in summer -highly unlikely!
Mild Fire.	Burn over 10% of reserve (no fences compromised).	Every 15 years.	0-3 months – 0.90 3-6 months – 1.00	0.90	Local	Occurs spring/autumn
Fox Incursion.	Fence cut. 1-2 foxes enter reserve –detected within 3 days, fox removed within 1 week.	Every 5 years	No impact	0.95	Local	Assumes checks from Tuesday - Thursday and rapid response.
Drought	Severe drought occurring for 3 years.	Every 15 years	No impact on breeding.	No impact on survival.	Global	Assumes irrigation and supplementary feeding.

*(proportion of normal values)

WORZ (Captive)

Note – in the model, Severe Fire is partitioned between Werribee and Melbourne such that only 1 institution is lost at a time.

Catastrophe	Nature of impact	Frequency	Impact severity* – reproduction	Impact severity* - survival	Local/Global	Notes/queries
Severe Fire.	Burn 100% of enclosures.	Every 50 years	0.00 (i.e. no reproduction)	0.00 (i.e. 100% loss)	Local	Only occurs in summer -highly unlikely!!!
Fox –small scale incursion – fence cut	NA	Every 5 years	No impact	0.05 (i.e. 0.95 loss of population)	Local	Assumes checks from Tuesday - Thursday and a rapid response
Drought	Severe drought occurring for 3 years.	Every 15 years	No impact on breeding	No impact on survival	Global	Assumes irrigation and supplementary feeding.

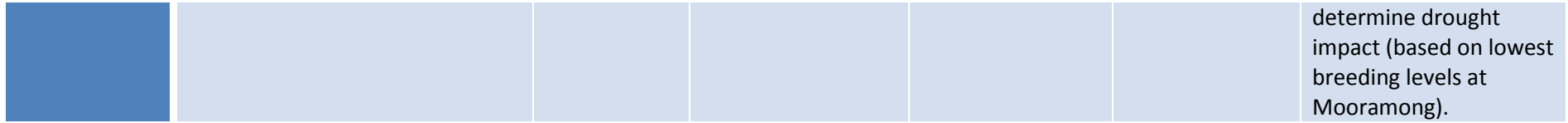
Melbourne Zoo (Captive)

Catastrophe	Nature of impact	Frequency	Impact severity* – reproduction	Impact severity* - survival	Local/Global	Notes/queries
Severe Fire	Fire on extreme weather day, caused by electrical short etc.	Every 50 years	0.00 reproduction	0.00	Local	Only occurs in summer -highly unlikely!!! (note only 1 fire in 150 years)
Drought	Severe drought occurring for 3 years.	Every 15 years	No impact on breeding	No impact on survival	Global	Assumes irrigation and supplementary feeding.

*(proportion of normal values)

Hypothetical Sites

Catastrophe	Nature of impact	Frequency	Impact severity* – reproduction	Impact severity* - survival	Local/Global	Notes/queries
Severe Fire	Burn over 90% of reserve and associated fox incursion. Fences reconstructed after 3 weeks and foxes removed.	Every 50 years	1.0 (no impact)	0.10 (i.e. 90% loss)	Local	Only occurs in summer.
Mild Fire	Burn over 33% of reserve (no fences compromised).	Every 15 years	0.90 (10% loss)	0.80 (i.e. 20% loss)	Local	Occurs spring/autumn.
Fox Incursion	1-2 foxes enter reserve –detected within 1 day, fox removed within 1 week.	Every 10 years	No impact	0.05 (i.e. 95% loss)	Local	Assumes at least daily checks and rapid response.
Drought	Severe drought occurring for 3 years.	Every 15 years	Summer: 0.1 Autumn: 0.1 Winter: 0.5 Spring: 0.5	0.90 (i.e. 10% loss)	Global	Climate change modelling required to update severity and frequency. Values may need adjustment. Look at data sets to



*(proportion of normal values)

Results and Discussion

Viability of individual release sites

Using the baseline parameters, with starting populations drawn from the customised file, and the catastrophe schedules described above, models were constructed for all wild, captive and hypothetical sites. The results are displayed in **Table 9**.

Table 9. Summary of results for all site-specific scenarios

Scenario	stoc-r	SD(r)	PE ₅₀	N-extant ₅₀	SD(Next) ₅₀	GeneDiv ₅₀	Median TE	Years TE
Hamilton	-0.011	0.250	1	0	0	0	67	16.75
Hypoth_Wild	-0.010	0.261	1	0	0	0	80	20
WORZ_Wild	0	0.241	0.900	30.70	24.85	0.5126	128	32
French_Island	0.034	0.222	0.716	3427.44	942.01	0.8698	97	24.25
Rothwell	0	0.254	0.704	102.23	84.56	0.6784	142	35.50
Woodlands	-0.015	0.262	0.848	141.41	115.80	0.6896	93	23.25
WORZ_Captive	0.137	0.558	0.994	53	51.29	0.2778	33	8.25
Melbourne_Captive	0.251	0.056	0.602	114.38	4.30	0.6469	153	38.25

As can be seen all sites have a high chance of extinction over the 50 year period (PE₅₀=0.70 – 1.00) for wild sites; 0.60– 0.99 for captive sites). This appears to reflect past experience at wild sites.

Further tests considered the impact of varying the carrying capacity at each site. For the purpose of the exercise, a “successful” scenario is one which shows:

- positive growth (i.e. mean stoc-r >0.00 over the 50 year period)
- probability of extinction less than 5% (i.e. PE₅₀ <0.05) in 50 years

Table 10: Population performance at each current and planned site with varied carrying capacity (K)

Scenario	stoc-r	SD(r)	PE ₅₀	N-extant ₅₀	SD (Next) ₅₀	GeneDiv ₅₀	Mean TE	Years TE
Hamilton_K70	-0.009	0.244	1	0	0	0	68.7	17.2
Hamilton_K100	-0.006	0.254	0.990	12.80	7.66	0.4702	84.3	21.1
Hamilton_K200	0.006	0.283	0.822	81.57	57.54	0.6453	89.2	22.3
Hamilton_K300	0.012	0.298	0.694	168.35	95.08	0.7323	88.8	22.2
Hamilton_K400	0.013	0.307	0.664	250.79	129.63	0.7613	86.1	21.5
Hamilton_K500	0.011	0.321	0.662	325.01	170.18	0.7667	84.9	21.2
Hamilton_K800	0.014	0.320	0.648	531.60	270.84	0.8032	81.5	20.4
Hamilton_K1000	0.011	0.329	0.654	612.49	374.43	0.7901	86.8	21.7
Hamilton_K2000	0.014	0.330	0.596	1177.76	779.00	0.8147	81.9	20.5
Rothwell_250	0.000	0.254	0.698	105.26	82.47	0.6423	103.5	25.9

Scenario	stoc-r	SD(r)	PE ₅₀	N-extant ₅₀	SD (Next) ₅₀	GeneDiv ₅₀	Mean TE	Years TE
Rothwell_K300	0.004	0.252	0.632	154.35	103.23	0.7011	103.8	26.0
Rothwell_K400	0.007	0.250	0.556	224.18	139.34	0.7315	103.7	25.9
Rothwell_K500	0.010	0.250	0.520	292.89	173.82	0.7645	90.7	22.7
Rothwell_K600	0.013	0.251	0.456	345.95	211.66	0.7874	91.6	22.9
Rothwell_K700	0.013	0.252	0.470	455.03	233.96	0.7929	85.0	21.3
Rothwell_K800	0.014	0.249	0.464	499.43	277.14	0.8009	81.0	20.3
Rothwell_K1000	0.015	0.248	0.436	624.98	341.39	0.8164	83.1	20.8
Rothwell_K2000	0.016	0.254	0.414	1206.68	704.87	0.8339	81.2	20.3
Woodlands_350	-0.016	0.260	0.832	116.89	110.43	0.6947	85.3	21.3
Woodlands_K500	-0.011	0.258	0.768	180.11	160.62	0.7190	83.0	20.8
Woodlands_K800	-0.008	0.257	0.712	334.63	275.36	0.7696	77.0	19.2
Woodlands_K1000	-0.007	0.258	0.710	472.70	337.58	0.7836	76.6	19.2
Woodlands_K2000	-0.006	0.257	0.702	967.32	731.06	0.8177	76.6	19.2
Hypoth_Wild_K100	-0.006	0.262	0.968	29.88	26.98	0.4879	101.5	25.4
Hypoth_Wild_K200	0.007	0.263	0.682	97.42	64.05	0.6653	102.9	25.7
Hypoth_Wild_K300	0.015	0.262	0.552	190.32	102.32	0.7264	99.5	24.9
Hypoth_Wild_K400	0.017	0.260	0.474	261.13	141.45	0.7529	98.7	24.7
Hypoth_Wild_K500	0.022	0.261	0.422	327.57	168.67	0.7778	96.7	24.2
Hypoth_Wild_K600	0.023	0.262	0.396	402.88	209.08	0.7882	81.3	20.3
Hypoth_Wild_K700	0.024	0.268	0.408	448.23	243.44	0.7954	88.9	22.2
Hypoth_Wild_K800	0.026	0.259	0.332	543.75	265.04	0.8014	91.8	23.0
Hypoth_Wild_K900	0.025	0.265	0.356	615.77	301.26	0.8103	84.2	21.1
Hypoth_Wild_K1000	0.029	0.259	0.290	719.56	327.68	0.8201	79.8	20.0
Hypoth_Wild_K1500	0.029	0.264	0.322	1077.74	490.39	0.8370	73.7	18.4
Hypoth_Wild_K2000	0.031	0.266	0.296	1398.86	660.78	0.8444	68.9	17.2

At currently estimated carrying capacities, none of the four wild sites show positive growth. All scenarios show high rates of extinction over 50 years (range PE₅₀ = 0.29 – 1.0) with mean time to extinction ranging from 17 – 25 years; that is, re-founding would be likely to be required at least once and often several times, during the 50-year period.

Modelling in the absence of catastrophes (see section on Sensitivity Testing) indicates that EBBs have the potential to achieve positive growth and extinction probabilities of less than 5% over 50 years, at K≥300. Neither Hamilton, WORZ nor Rothwell are large enough to achieve this, even if it were possible to remove catastrophes. Woodlands is the only site that could meet this capacity requirement but is currently hampered by the frequency and severity of catastrophes.

French Island and Cats

In the absence of cats, the French Island model shows positive growth (around 7% per time-step), large mean population size (N=3630.44) but a high probability of extinction (PE₅₀= 0.62; with mean Time to Extinction=20.28 years). Doubling the founder base to 120 individuals makes little difference

to the likelihood of extinction, but does increase average gene diversity (from 85.94% to 90.20% at 50 years).

In the presence of cats, growth is negative, mean times to extinction are considerably lower, and probability of extinction increases to between 94 and 100%. Of the three scenarios modelled (low impact, high impact (acting uniformly across age-classes) and high impact (operating selectively across age-classes), the “low impact” shows some potential for growth but even there, due to increased vulnerability, the population is unable to make use of the large capacity of the island, with mean size generally low and highly variable (Mean N = 716.78, S.D. 1033.03). The effects are not ameliorated by doubling founder input to 120 individuals, nor by entirely removing catastrophes from the island, though the latter brings some improvements.

It should be noted that limited information was available on the impact of cats and so the effects are modelled uniformly across the population. In reality a lack of uniformity of effect could improve performance in the presence of cats but more data would be required to test this.

Table 11: Summary of results for French Island scenario testing. Scenarios highlighted in red averaged negative growth over the period.

Scenario	stoc-r	SD(r)	PE ₅₀	N-extant ₅₀	SD(Next) ₅₀	GeneDiv ₅₀	MeanTE	Years TE
FI_No_Cats	0.071	0.242	0.620	3620.44	745.04	0.8594	81.1	20.3
FI_120fdrs	0.071	0.240	0.632	3580.17	732.71	0.9020	78.6	19.7
FI_K800_No_Cats	0.067	0.246	0.662	700.39	178.04	0.8117	83.5	20.9
FI_Low_Impact_Cats	-0.021	0.254	0.936	716.78	1033.03	0.5304	57.5	14.34
FI_High_Uniform_Cats	-0.134	0.274	1	0	0	0	17.7	4.4
FI_High_Selective_Cats	-0.063	0.264	0.998	4	0	0.6563	34.9	8.7
FI_No_Catastrophes	0.080	0.200	0.002	3819.87	321.46	0.8825	11.0	2.8
FI_Low_Impact_No_Catastrophes	-0.009	0.213	0.666	1056.49	1198.11	0.6259	66.1	16.5
FI_High_Uniform_No_Catastrophes	-0.118	0.252	1	0	0	0	21.3	5.3
FI_High_Selective_No_Catastrophes	-0.053	0.237	0.988	65.17	52.82	0.5484	45.5	11.4
FI_Low_Impact_120fdrs	-0.017	0.245	0.896	774.48	872.27	0.6926	68.3	17.1
FI_High_Uniform_120fdrs	-0.126	0.260	1	0	0	0	23.7	5.9
FI_High_Selective_120fdrs	-0.060	0.253	1	0	0	0	47.0	11.8

Figure 9. shows a comparison of mean population sizes over time, for populations surviving the 50 year period.

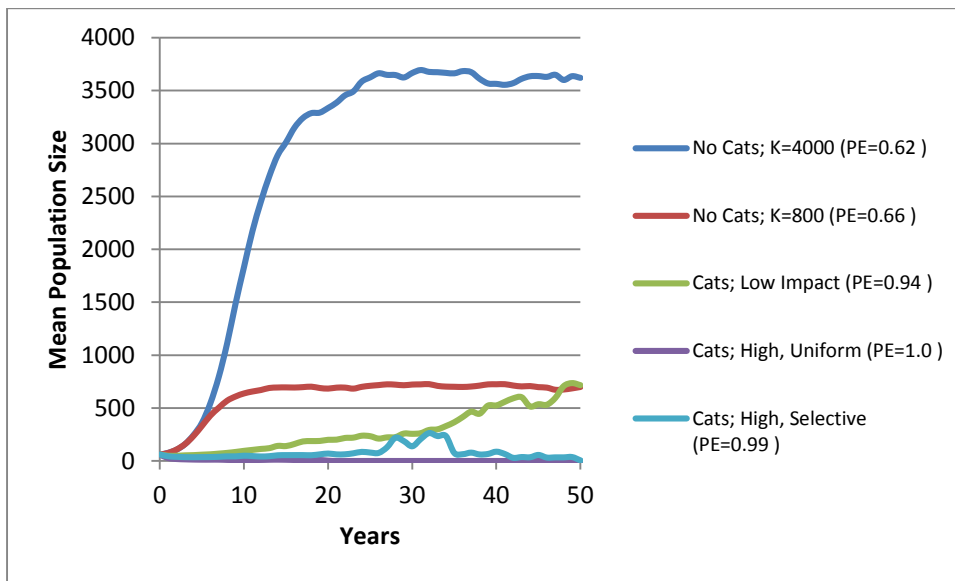


Fig 9: Mean population size over time for populations surviving the 50-year period. Note the high extinction probability in all cases.

Models were also run for sites similar to French Island in all respects, including the catastrophe regime, but without cats and at carrying capacity of $K=800$. The smaller site outperformed all of the cat scenarios considered and in terms of growth, extinction risk and mean time to extinction, performed almost as well as for $K=4000$, though the latter retained considerably more gene diversity at 50 years (85.94%GD for $K=4000$; 81.17%GD for $K=800$).

The impact of supplementation at smaller sites

All of the smaller sites are vulnerable to extinction over the 50-year time period. The impact of supplementation on extinction trajectory was explored by supplementing these populations at different time intervals (5, 10, 15, 20 and 25 years) and with different numbers of animals (8 and 15 founder adults, equal sex-ratio). Results are presented below.

The impact of regular supplementation at Hamilton would be to prop up (rather than “rescue” an otherwise permanently declining and vulnerable population. From **Figures 10** and **11** we can see that an input of 8 animals every 5 years starts to place the population on a relatively even keel, though population sizes tend to be low with respect to carrying capacity (Mean $N=39.73$; $K=70$). Extending the time interval to 10 years results in a series of steep population declines and generally low population sizes (Mean $N=17.39$).

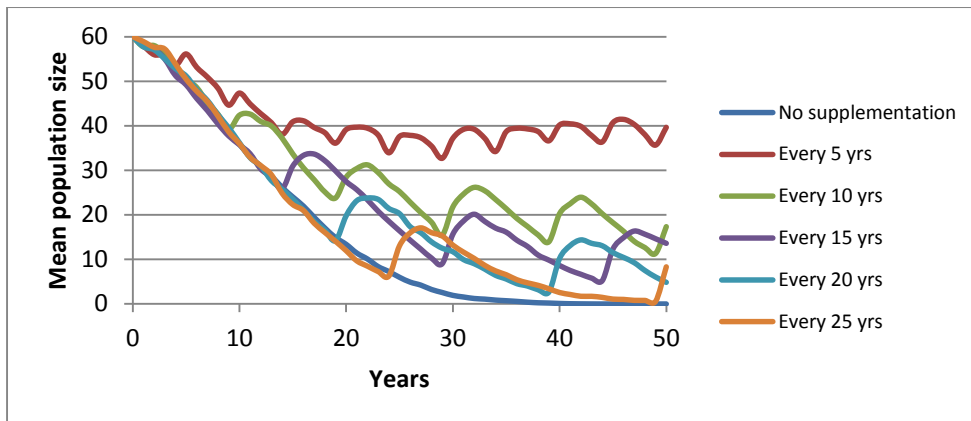


Figure 10: Mean population size over time at Hamilton, under a supplementation rate of 8 individuals at varied time intervals.

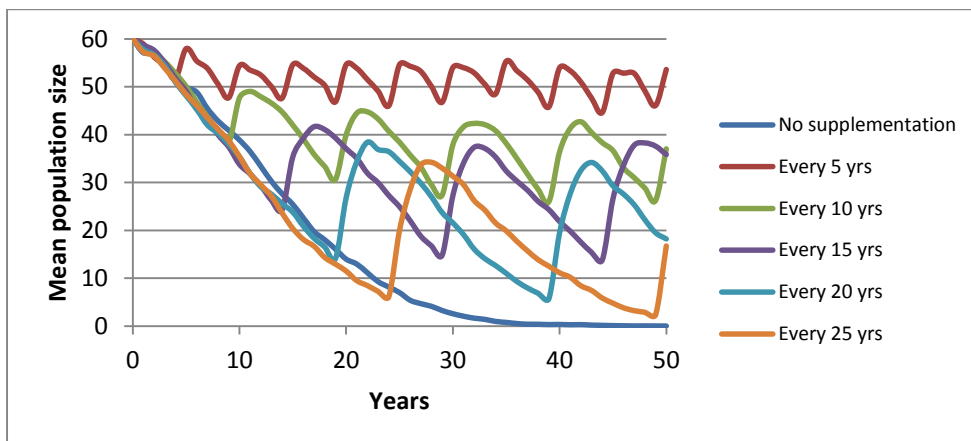


Figure 11: Mean population size over time at Hamilton, under a supplementation rate of 15 individuals at varied time intervals.

As might be expected, increasing the number of animals added at each supplementation event, to 15, improves average growth rates and population sizes. At 5 year supplementation intervals, mean population size increases from $N=39.63$ to $N=53.68$, though this is still below the estimated carrying capacity of $K=70$. Though mean growth rate is positive at 5, 10 and 15 year intervals, dramatic fluctuations in size can be seen that coincide with distance from supplementation events.

Similar patterns are seen in the other site models, though the larger carrying capacities and in some cases more moderate catastrophe schedules, allow for less erratic results than those seen for Hamilton. Supplementing at a rate of 8 animals per event generally provides for positive growth and steady population sizes only when applied at 5 year intervals. In these cases, population sizes tend to persist at a level below carrying capacity.

Increasing the supplementation rate to 15 animals per event generally sustains population sizes closer to capacity where applied at 5 year intervals, and this is also true at WORZ and Rothwell (though not for Woodlands) when the interval is extended to 10 years. **Tables 12** and **13** show results for scenarios in which positive growth was achieved over the 50-year period.

Other supplementation graphs and a full table of results for wild sites, are provided in **Appendix IV**.

Table 12: Supplementing with 8 individuals (only scenarios in which mean population growth was positive are included).

Scenario	stoc-r	SD(r)	PE ₅₀	N-all ₅₀	SD(Nall) ₅₀	GeneDiv ₅₀	MeanTE
Hamilton							
At 5yr intervals	0.020	0.259	0	39.63	23.65	0.8405	82.2
At 10yr intervals	0.001	0.257	0	17.34	17.09	0.8577	87.3
WORZ							
At 5 yr intervals	0.029	0.247	0	73.61	32.62	0.8314	107.6
At 10yr intervals	0.014	0.246	0	48.92	38.05	0.8153	104.9
At 15yr intervals	0.010	0.246	0.210	37.58	35.85	0.7224	113.9
At 20yr intervals	0.005	0.246	0.522	23.20	34.00	0.6521	110.3
At 25yr intervals	0.004	0.245	0	18.84	23.74	0.8486	113.7
No supplementation	0.000	0.239	0.904	2.84	11.18	0.4741	116.5
Mount Rothwell							
At 5 yr intervals	0.021	0.258	0	149.24	86.04	0.8382	99.2
At 10yr intervals	0.006	0.260	0	85.77	90.91	0.8325	93.2
At 15yr intervals	0.006	0.256	0.250	74.70	87.38	0.7437	96.0
At 20yr intervals	0.003	0.256	0.476	55.50	78.29	0.7137	103.8
At 25yr intervals	0.002	0.254	0	52.90	78.09	0.8376	105.4
No supplementation	0.001	0.251	0.712	33.22	70.07	0.6793	106.4
Woodlands							
At 5 yr intervals	0.004	0.270	0	128.89	122.34	0.8464	91.5

Table 12: Supplementing with 15 individuals (only scenarios in which mean population growth was positive are included)

Scenario	stoc-r	SD(r)	PE ₅₀	N-all ₅₀	SD(Nall) ₅₀	GeneDiv ₅₀	MeanTE
Hamilton							
At 5 yr intervals	0.038	0.259	0	53.68	20.29	0.8791	81.0
At 10yr intervals	0.015	0.258	0	37.60	22.36	0.8728	78.6
At 15yr intervals	0.008	0.256	0.15	35.37	24.90	0.7876	77.5
At 20yr intervals	0	0.254	0.45	19.33	23.61	0.7062	74.9
WORZ							
At 5 yr intervals	0.043	0.249	0	89.68	25.74	0.8729	106.6
At 10yr intervals	0.026	0.248	0	72.54	35.27	0.8420	105.6
At 15yr intervals	0.019	0.244	0.04	67.77	35.38	0.7924	101.9
At 20yr intervals	0.014	0.247	0.19	54.57	38.81	0.7363	105.1
At 25yr intervals	0.009	0.244	0	34.47	29.51	0.8633	111.9
Mount Rothwell							
At 5 yr intervals	0.030	0.258	0	182.90	76.39	0.8751	87.3
At 10yr intervals	0.017	0.259	0	139.60	90.20	0.8475	88.5
At 15yr intervals	0.013	0.257	0.05	123.00	88.16	0.7993	83.3

Scenario	stoc-r	SD(r)	PE ₅₀	N-all ₅₀	SD(Nall) ₅₀	GeneDiv ₅₀	MeanTE
At 20yr intervals	0.006	0.259	0.25	95.22	95.32	0.7601	92.2
At 25yr intervals	0.004	0.261	0	68.36	80.86	0.8628	102.8
No supplementation	0	0.252	0.70	34.72	72.55	0.6534	109.6
Woodlands							
At 5 yr intervals	0.017	0.267	0	210.50	115.8	0.8753	79.1
At 10yr intervals	0	0.270	0	121.40	116.84	0.8600	88.7

Note that probabilities of extinction may be counter-intuitive in some cases. For example PE for 25 year supplementation scenarios may be lower than that for more frequent supplementation. This relates to the year of last supplementation and its distance from year 50. So, for example, supplementing at 25 years will place the final year of supplementation at year 50, so that “empty” sites that would otherwise have been counted as “Extinct” are counted as “Extant”, thereby over-stating the viability of the scenario.

How big might a mainland meta-population of EBBs need to be and when?

The IUCN/SSC advocates a range-wide, species-level approach to conservation planning wherever possible. This is particularly important when setting target population sizes and would ideally be the approach taken here; that is, any minimum target size set for mainland EBBs should take into account estimates of size, trends and management for Tasmanian counterparts. In the short-term this may be difficult to arrange due to geographical and administrative separation and interim targets for the mainland component may need to be agreed in isolation.

Based on a PVA meta-analysis of population viability analyses, Traill *et al.* (2007) advocate minimum sizes of several thousands (95% CI=3577-5129) as a general rule of thumb. This is not dissimilar from the MVP for long-term genetic security suggested by Franklin (1980), of a genetically effective size of 500, which in wild populations is expected to require a census population of around 5000 (Frankham, 1995c) and relates to an estimated “mutation-drift balance”, at which populations are expected to accumulate as much new diversity through mutation as they lose through drift (chance processes). In the case of mainland EBBs, which are a component of a wider taxon, a smaller size can be appropriate, in the context of occasional gene-flow across the broader taxon, and a target size of about half of this (N=2500) has been suggested (Weeks, internal report). The modelling results, which incorporate demographic uncertainty, confirm relatively high performance both genetically and demographically, at mean sizes of around N=1500-2500 (see **Table 13**), though scenarios achieving these mean sizes have much larger carrying capacities (K≈4000-8000) and show considerable fluctuation about the mean. In short, pending a range-wide, species-level review of this target at some future point, the models support the proposed interim target of 2500, though indicate that this may require a carrying capacity of 4000-8000.

Population growth towards target size is required urgently to slow further genetic erosion, which will proceed rapidly at current population sizes and given the species’ short generation time.

How should a meta-population be structured?

As described previously, three alternative management strategies were compared. These were:

- 1) NOW – this models a meta-population comprising all current and proposed sites, excluding French Island (that is, Hamilton, Mount Rothwell, Woodlands and WORZ).
- 2) HYPOTHETICAL SITE Additions – this models a meta-population comprising the NOW scenario with the addition of 1-4 medium-sized sites of K=500 and K=800.

- 3) BIG SITE Additions – this models a meta-population that adds, to the NOW scenario, first one and then two, large sites on the scale of French Island, but in the absence of cats.

All sites are re-founded using captive animals when they go extinct.

Table 13: Relative performance of alternative meta-population management strategies for EBBs

Scenario	Total K	Stoc-r	SD(r)	N-all	SD(Nall)	GD ₅₀	SD(GD ₅₀)	Allele N ₅₀	SD(Allele N ₅₀)	F ₅₀
Metapop Now	775	0.02	0.182	221.62	125.31	0.7929	0.0771	9.67	2.75	0.2913
1 New Site K500	1275	0.025	0.174	432.16	236.82	0.8185	0.0683	11.57	3.29	0.2701
2 New Site K500	1775	0.027	0.168	634.87	306.9	0.8376	0.0602	12.87	3.39	0.2605
3 New Site K500	2275	0.029	0.164	821.12	365.04	0.8405	0.0544	13.23	3.11	0.2628
4 New Site K500	2775	0.029	0.162	998.48	455.1	0.8416	0.0501	13.27	3.14	0.2612
1 New Site K800	1575	0.027	0.177	600.14	333.93	0.825	0.0726	12.22	3.28	0.2553
2 New Site K800	2375	0.029	0.172	918.39	471.37	0.837	0.0623	13.05	3.25	0.2454
3 New Site K800	3175	0.031	0.168	1214.2	587.82	0.844	0.0555	13.78	3.21	0.2434
4 New Site K800	3975	0.031	0.166	1427.04	649.21	0.8926	0.0431	23.43	7.14	0.2482
1 Big Site	4775	0.028	0.221	1657.98	1646.73	0.8601	0.0714	20.32	10.6	0.2277
2 Big Sites	8775	0.033	0.204	3047.45	2464.05	0.8783	0.0678	25.31	12.79	0.2026

As all sites are re-founded from captivity following an extinction event and as the captive population modelled is able to cope with the required rate of re-founding in all scenarios, the probability of meta-population extinction is zero in all cases.

Table 13 shows a range of demographic and genetic performance indicators for the scenarios modelled. Carrying capacities vary from K=775 to K=8775. Results indicate that numbers on the ground can vary dramatically over time within each scenario and this is illustrated in **Figures 12** and **13**, which show ten iterations of the least successful (NOW) scenario and of the most successful (NOW plus 2 BIG SITES each at K=4000). Note though that the summary figures presented may exaggerate this variation somewhat due to the late start and long growth phase of some of the individual site populations. Stochastic growth is relatively constant across scenarios, at around 2-3% each year.

Figure 12: Snapshot after 10 iterations of scenario NOW.

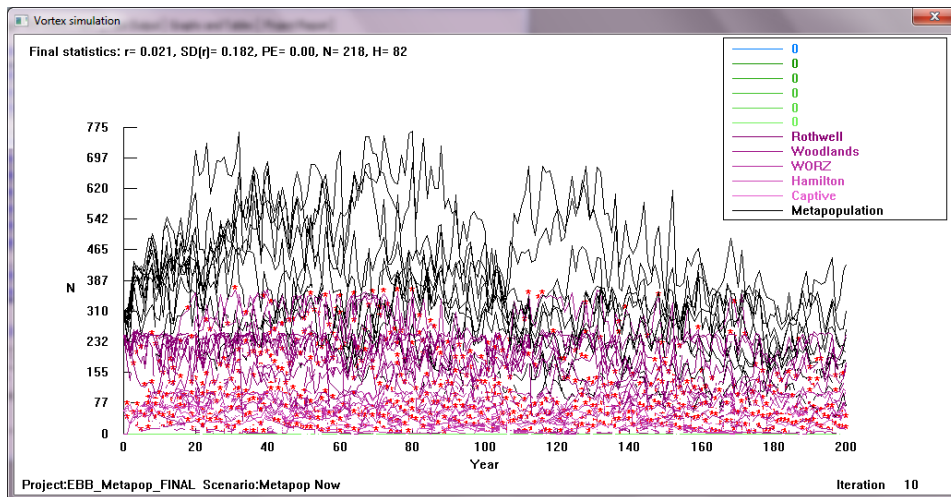
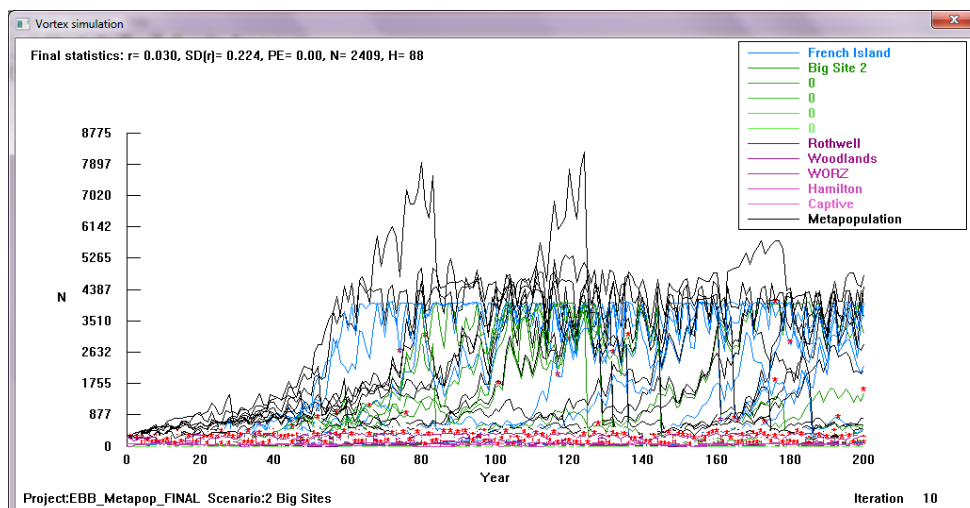
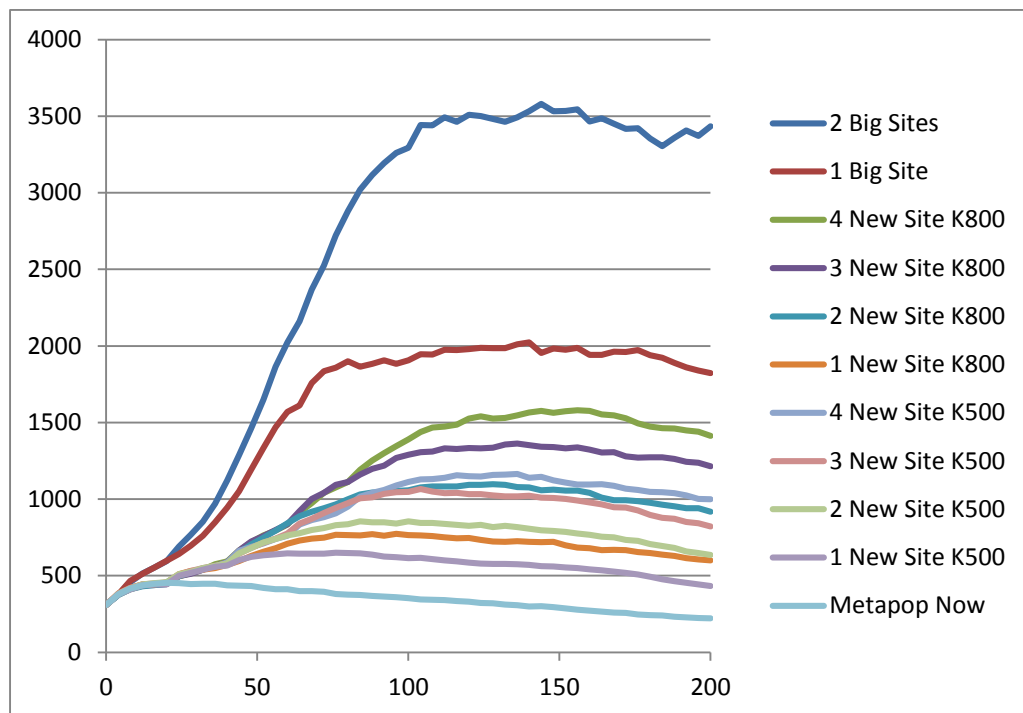


Figure 13: Snapshot after 10 iterations of scenario NOW plus 2 BIG SITES each at K=4000



As can be seen, mobilising all currently available sites (Hamilton, Rothwell, Woodlands and Werribee Open Range Zoo (WORZ)) but excluding French Island, performs the least well both demographically and genetically of all scenarios modelled, whilst the 2 Big Sites scenario (current plans plus the addition of two French Island-sized sites, without the influence of cats) performs best. This might be predicted on the basis of carrying capacity (and resulting population sizes – see **Figure 14**) alone, as these scenarios sit at opposing ends of the available range. However, there are some more subtle effects occurring that relate not just to total carrying capacity but to meta-population configuration.

Figure 14. Mean population size over time for meta-population scenarios



The scenario which includes 4 New Sites at K=800 (total K=3975; GD=0.89; Alleles retained=23.43) performs better in terms of gene diversity retention than the 1 Big Site scenario (total K=4775; GD=0.86; Alleles retained=20.32) and indeed only slightly less well than the very large 2 Big Sites scenario (total K=8775; GD=0.88; Alleles retained=25.31). Population genetics theory predicts that population sub-division will give rise to greater overall allelic diversity through the differential fixation of alleles as within-subdivision homozygosity increases – provided that the accompanying loss of fitness within sub-divisions does not result in their loss. This is likely to be a contributor to the success of the 4 Sites at K=800 scenario.

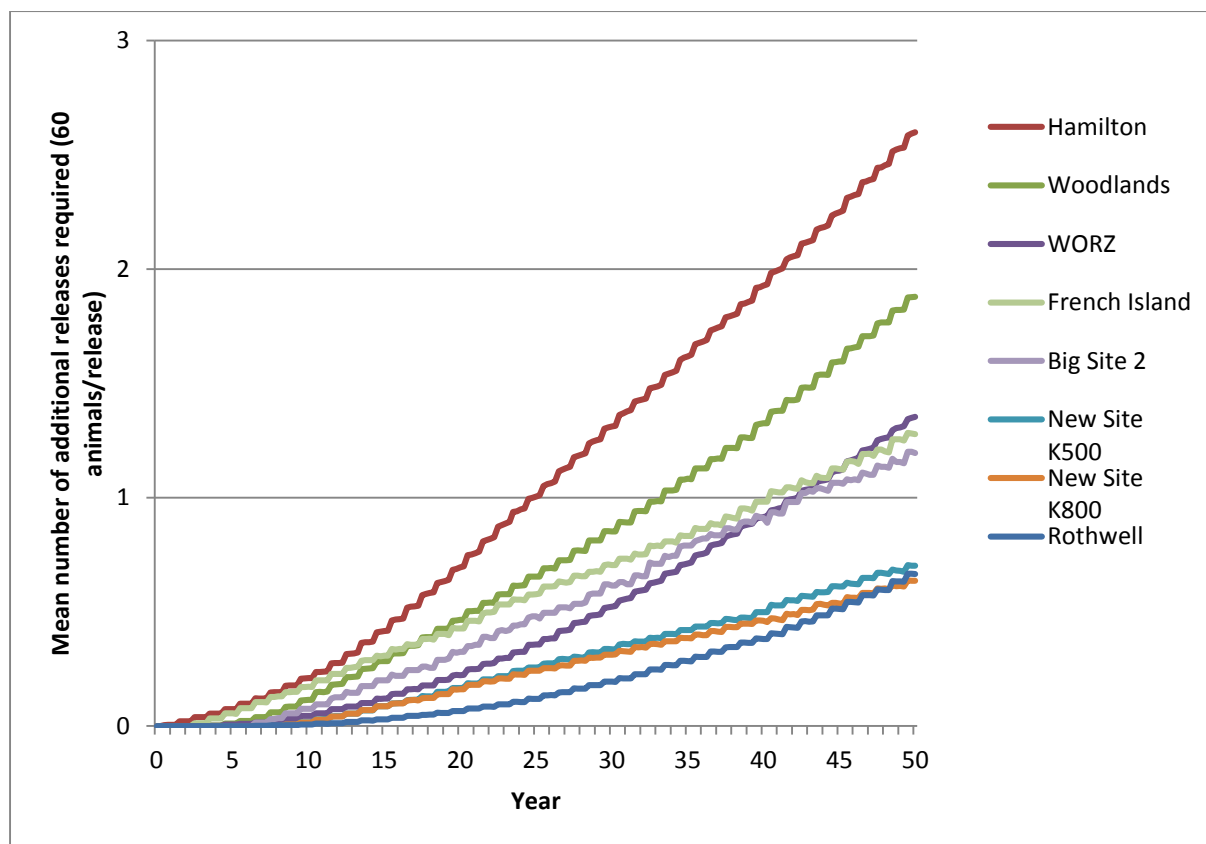
With the exception of the current scenario (Metapop NOW), which performs particularly poorly, all other scenarios perform similarly with respect to gene diversity retention and at a level markedly below the 3 highest performing scenarios described above.

Inbreeding accumulation is high across all scenarios modelled, with the coefficient at 50 years ranging from $F=0.20$ – 0.29 . Full-sibling matings are expected to produce an F value of 0.25 and detrimental effects can be detected in most captive populations at this level. The Big Site populations show lower average inbreeding than the more sub-divided sites at 50 years due to the within-population inbreeding accumulation that would occur in the absence of movements between sub-divisions. Regular translocation between sub-divisions of K=500 or K=800 would be expected to reduce inbreeding, though too much movement may reduce the beneficial impacts of sub-division on allelic diversity. Optimal translocation strategies might be a useful focus for further modelling, though additional information on inbreeding susceptibility (some purging is likely to have occurred and this can be factored into the models), genetic make-up of remaining EBBs and post-translocation survival and integration would ideally inform this. More dramatic results with respect to reducing

inbreeding and enhancing gene diversity would be expected from the translocation of Tasmanian animals into the mainland population, however, rapid population expansion would be needed to secure these benefits, which would erode swiftly at the population's current size.

Figure 15 illustrates, for each sub-population, the mean number of additional releases required each year (i.e. post founding phase), as an indicator of relative management intensity required. Not surprisingly, the smaller and the more catastrophe-prone sites require more regular re-founding, with Hamilton and Woodlands expected to require around 2-3 additional re-founding events, WORZ and French Island (without cats) requiring around 1, and Rothwell and the Hypothetical New Sites (at K=500 or 800) typically requiring less than 1 additional re-founding event, over the 50-year period.

Figure 15. Additional releases over time



In summary, of the meta-population strategies modelled, the one which performs least well is that which includes only the smaller sites (Hamilton, Woodlands, Rothwell and WORZ). This requires the highest number of re-founding events and results in particularly low values for gene diversity and particularly high levels of inbreeding (though active management could reduce the latter to some extent). Extinctions can be reversed through re-founding, and in the models this is all done from a relatively secure captive population. In order to harvest for re-founding from wild populations without placing them at risk, it would be important to understand both the carrying capacity of each wild site and also what proportion of that is currently occupied. Models indicate that the margin of error for this in the smaller sites is relatively slight and advice from participants is that the difficulty of measuring these parameters is high; taking a precautionary approach might therefore preclude any exit strategy for the captive population as a source of release animals, in this meta-population scenario.

Adding French Island to the mix improves the situation considerably, but only if the impact of cats proves negligible or significantly less uniform than is currently described in the models. Adding instead four sites at $K=800$ performs as well and in some areas better than this, for less overall carrying capacity. Only the addition of two French Island-like sites exceeds this success, but it does so only slightly, requires more than double the carrying capacity to do so, and is arguably somewhat less effective at spreading risk. On the basis of the tests carried out then, a meta-population strategy that includes four sites of around $K=800$, in addition to the smaller sites already in play, provides the best all-round results. *[Note that there may be other recovery objectives (e.g. cost, community expectation) not considered here that would increase the value of other strategies; this would need to be discussed by a broader group].*

Implications for the captive program

The length of time over which the captive population is required for release and/or insurance will depend on the choice of meta-population strategy and the rate of identification and preparation of wild release sites.

If one or other of the smaller scenarios is selected (e.g. NOW, or NOW plus one other site) a precautionary approach would see the captive population as a permanent fixture within the meta-population, both for ongoing insurance and as a periodic source of animals for release as wild populations fail. If one or other of the larger scenarios are selected and fully realised (e.g. NOW plus 4 sites at $K=800$ or NOW plus 1-2 Big Sites), the rate of re-founding required should be able to be met in an ongoing way through harvest from well-populated wild sites. In these scenarios, the captive population could play a short-term release role whilst sites are identified and seeded, followed by an insurance role while they become established, and beyond that could be wound-down and its role(s) replaced by other wild populations. Exact time periods are difficult to assess without additional information. In the models, new sites are founded at 5-year intervals, which stretches the site founding phase out to two decades in some cases. However, the modelled captive population can cope with a more aggressive harvesting regime if required. The length of time for which the captive population would be needed in the larger scenarios then, is likely to be determined by the rate of identification and preparation of wild sites.

In all of the meta-population scenarios modelled there is a release-intensive initial phase followed by periods of years in which no releases are required. This poses a challenge for the captive program, which must be able to produce large numbers of “surplus” animals in some years and none in others, and must do so without risking demographic crashes due to reproductive senescence or genetic deterioration resulting from regular bottlenecks. A flexible population control method such as pouch management is likely to be required, ideally in combination with a meta-population founding strategy that allows for a shorter but more intensive period of wild population establishment, should this be achievable .

Refining the models

It should be noted that these general conclusions have been drawn based on the parameters and estimates agreed during the workshop. There was considerable uncertainty attached to some of the parameters and those that have the greatest influence on population performance are as follows (in order of importance):

- site-specific frequency and severity of catastrophes
- site-specific carrying capacities
- age-specific mortality
- average percentage of females breeding in each 3-month time-step
- litter size in each 3-month time-step
- inbreeding severity and, informing this, starting genetic make-up and purging history

Further clarity around these would be expected to improve the value of the models.

Next steps

The analyses presented here discuss the relative performance of individual sites – some real and some hypothetical – and of a range of predominantly hypothetical meta-population management scenarios. The value of this, it is hoped, will be in:

- identifying and quantifying some additional characteristics of a site that might predispose it to success;
- informing the search for new, potentially valuable sites;
- developing new meta-population management scenarios for testing, based on these real sites.

Logical next steps in this analysis would involve the identification of new, real scenarios that can be represented in the models and, with additional information relating to cost, site management difficulty and other relevant factors, can be compared in a transparent way using a list of objectives agreed and if necessary weighted, by decision-makers. **Table 14** below illustrates an example of this kind of analysis for the meta-population scenarios considered here, showing the relatively high performance of the NOW plus 4 sites of K=800. *[Note that some of the objectives used in this example were inferred from discussions and were not formally agreed or weighted by participants].*

Table 14. Example comparison of site performance across a range of objectives (top three performers are highlighted for each objective).

	Measure	NOW	Extra sites of K=4000		Extra sites of K=500				Extra sites of K=800			
			NOW+1	NOW+2	NOW+1	NOW+2	NOW+3	NOW+4	NOW+1	NOW+2	NOW+3	NOW+4
DRAFT Objectives												
High Gene Diversity	Prop. Het	0.79	0.86	0.88	0.82	0.84	0.84	0.84	0.83	0.84	0.84	0.89
High Allelic Diversity	Alleles _{N50}	9.67	20.32	25.31	11.57	12.87	12.23	13.27	12.22	13.05	13.78	23.43
Low Inbreeding	F ₅₀	0.29	0.23	0.20	0.27	0.26	0.26	0.26	0.26	0.25	0.24	0.25
Mgmt intensity (1)	Release effort/site	125	120	117	111	101	95	90	110	100	92	87.5
Mgmt intensity (2)	Easier(?) or Harder	Easier	Harder	Harder	Easier	Easier	Easier	Easier	Easier?	Easier?	Easier?	Easier?
Exit for captive pop	YES(?)/NO	No	Yes?	Yes	Yes?	Yes?	Yes	Yes	No	Yes?	Yes	Yes
Low risk of loss	No. pops at K>300	1	2	3	2	3	4	5	2	3	4	5

Summary

In summary, the inherent volatility of EBBs and the environment which they inhabit, argues for a meta-population strategy in which individual populations show some resilience to genetic and demographic stochasticity (i.e. site K is at least 300 and ideally larger) and where the expected frequency and impact of catastrophes are not prohibitively high. Risk of extinction is ideally spread

across several populations which are unlikely to be affected by the same catastrophe simultaneously. The strategy tested in this exercise that performed particularly well for a relatively modest boost in capacity, was that of adding to currently occupied and planned sites (Hamilton, Woodlands, Rothwell and WORZ), four additional sites each at $K=800$. In general, meta-population carrying capacities of $K= 4,000-8,000$ performed well both genetically and demographically and this might usefully be considered in the context of setting a target carrying capacity size for mainland EBBs, though ideally this would be done as part of larger, range-wide, species-level planning exercise incorporating Tasmanian stocks. Immediate population expansion is required to slow genetic deterioration and careful and innovative captive program design will be critical to the success of the project.

The models developed for this analysis are stored at CBSG and can be mobilised and modified as required to assist with future management planning. Logical next steps in this analysis would involve the identification of new, real scenarios that can be represented in the models and, with additional information relating to cost, site management difficulty and other relevant factors, can be compared in a transparent way using a list of objectives agreed and if necessary weighted, by decision-makers

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Appendix I: Workshop Participant Details

Name	Organisation	Role	Email
Peter Courtney	Zoos Victoria (Melbourne Zoo)	Curator, Threatened Species; EBB Species Coordinator	pcourtney@zoo.org.au
Graeme Coulson	University of Melbourne	Assoc Prof, Department of Zoology	gcoulson@unimelb.edu.au
Dan Harley	Zoos Victoria	Threatened Species Biologist	dharley@zoo.org.au
Michael Magrath	Zoos Victoria	Senior Scientist	mmagrath@zoo.org.au
Marissa Parrott	Zoos Victoria	Reproductive Biologist	mparrott@zoo.org.au
Charles Todd	Arthur Rylah Institute, DSE	Senior Scientist, Ecological Modelling	charles.todd@dse.vic.gov.au
Andrew Weeks	Centre for Environmental Stress and Adaptation	Director and Research Geneticist	aweeks@cesaraustralia.com aweeks@unimelb.edu.au
Madelon Willemsen	Zoos Victoria (Werribee Open Range Zoo)	Curator	mwillemsen@zoo.org.au
Amy Winnard	University of Melbourne	Specialist in the Reintroduction Biology of EBB	amy.winnard@unimelb.edu.au

Appendix II: Workshop Agenda

Conservation Planning Workshop for Eastern Barred Bandicoot (*Perameles gunnii*)

Hosted by Zoos Victoria
September 30 – October 2, 2012

Introduction

Zoos Victoria has invited the IUCN/SSC Conservation Breeding Specialist Group to facilitate a three-day conservation planning workshop for eastern barred bandicoots (*Perameles gunnii*). The workshop will be held in the Zoos Victoria Board Room from September 30th to October 2nd, 2012.

Eastern barred bandicoots have been the subject of conservation planning for several decades; recovery plans are already in place, as are reviews of previous successes and failures. The forthcoming workshop aims to build on and support these efforts.

Discussions will focus on developing the detail around a long-term strategy for the genetic and demographic management of remaining and proposed populations of the species. Amongst other things, we expect the workshop to consider:

- minimum target population sizes for specific sites and for the broader meta-population;
- optimal rates of exchange between sites;
- strategies for maximising gene diversity retention;
- roles and optimal parameters for the intensively managed captive population.

The principal workshop organiser is Marissa Parrott of Zoos Victoria: mparrott@zoo.org.au. CBSG workshop facilitators will be Kathy Traylor-Holzer: kathy@cbsg.org; and Caroline Lees: caroline@cbsgaustralasia.org.

Invitees

Richard Hill, Chair, Bandicoot Recovery Team, Department of Sustainability and Environment (DSE):

Peter Courtney, Curator, Threatened Species, Melbourne Zoo: pcourtney@zoo.org.au

Graeme Coulson, Assoc Prof, Department of Zoology, University of Melbourne: gcoulson@unimelb.edu.au

Dan Harley, Threatened Species Biologist, Zoos Victoria: धारley@zoo.org.au

Michael Kidman, Senior Keeper, Werribee Open Range Zoo: mkidman@zoo.org.au

Michael Magrath, Senior Scientist, Zoos Victoria:

Alan Robley, Senior Scientist, Arthur Rylah Institute, DSE: alan.robley@dse.vic.gov.au

Charles Todd, Senior Scientist, Ecological Modelling, Arthur Rylah Institute, DSE:

Andrew Weeks, Director, Centre for Environmental Stress and Adaptation Research, Geneticist: aweeks@cesaraustralia.com;

Madelon Willemsen, Curator, Werribee Open Range Zoo:

Amy Winnard, Specialist in the Reintroduction Biology of EBBs: amy.winnard@unimelb.edu.au

DRAFT Agenda

Day 1:

- 10.30am Welcome and introductions
- 11.00am Introduction to the workshop
- 11.15am Scene-setting presentations
- 12.30 LUNCH**
- 1.00pm Vision, goals and targets
- 2.30pm Vortex simulation models
- 3.00pm Model progress so far
- 3.30pm TEA BREAK**
- 3.45pm Review of model parameters
- 5.00pm End of DAY 1**

DAY 2:

- 9.00am Presentations
- 9.30am Developing management scenarios (working group)
Testing models (working group)
- 10.30am TEA BREAK**
- 10.45am Developing management scenarios (working group)
Testing models (working group)
- 12.30am LUNCH**
- 1.30am Developing management scenarios (working group)
Testing models (working group)
- 3.00pm TEA BREAK**
- 3.15pm Working group reports and discussion
- 4.30pm End of DAY 2**

DAY 3:

9.00am	Introduction to DAY 3
9.30am	Testing the consequences of management scenarios (working group) Collating EBB research and identifying knowledge gaps (working group)
10.45am	TEA BREAK
11.00am	Testing the consequences of management scenarios (working group) Collating EBB research and identifying knowledge gaps (working group)
12.30pm	LUNCH
1.30pm	Testing the consequences of management scenarios (working group) Collating EBB research and identifying knowledge gaps (working group)
3.00pm	TEA BREAK
3.15pm	Working group reports and discussion
3.45pm	Next steps
4.30pm	Close of workshop

Appendix III: Wild Growth Rates

The following formula was used to calculate lambda per time-step from observed growth:

$$(N_t/N_0)^{1/\text{time-steps}}$$

As many of the populations studied were studied during periods of decline, representative growth rates have been difficult to estimate. Plausible rates of growth in released populations were inferred as follows:

Woodlands

- 1989-1991 - 62 bandicoots released.
- By 1992 - population estimate was 90, 24 animals were added (bringing the total to 114?) (recorded in Winnard, 2010).
- By 1993 population size was at least 321 and possibly in excess of 500 (cited in Todd, 2002).
- Seebeck 1999 (cited in Winnard 2010) suggests the population reached its peak in 1994/95 at over 600 animals.

This suggests an increase of 114 to 321 or 114 to 500+ in approximately 1 year, or 4 time-steps.

- 114 – 321 in 1 year ($\lambda=2.815$) (λ per time-step = 1.29)
- 114-500 in 1 year ($\lambda=4.39$) (λ per time-step = 1.29)
- 321-600+ in 1 year ($\lambda=1.97$) (λ per time-step = 1.16)
- 500-600+ in 1 year ($\lambda=1.2$) (λ per time-step = 1.04)

Hamilton

Similarly at Hamilton, Winnard, 2010 records 20 individuals trapped in 1990, and a population estimate in 1993 of 90. That is an estimated increase of 70 individuals over 3 yrs ($\lambda = 1.52$) or λ per time-step = 1.11

Summary

From the information available at the workshop a plausible range for wild growth per time-step, was estimated as **1.04 – 1.29**

Appendix IV: Supplementation Trials

Results of supplementing populations at 5, 10, 15, 20 and 25 year intervals, with 8 and 15 additional animals, are provided here.

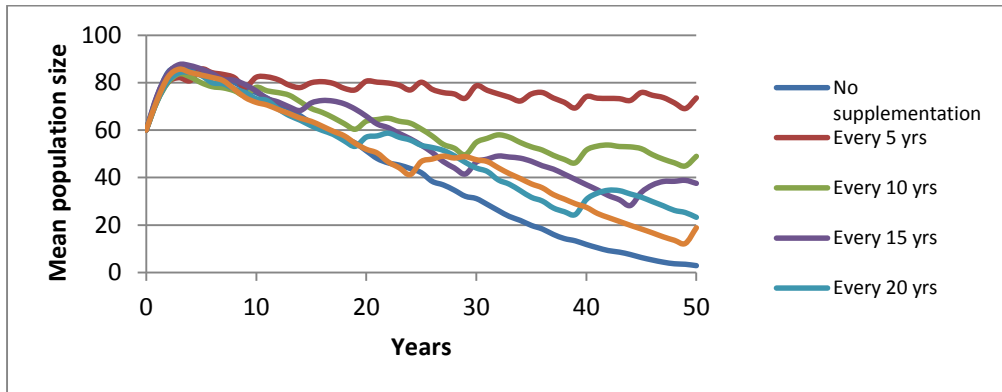


Figure A: Mean population size over time at WORZ, under a supplementation rate of 8 individuals at varied time intervals.

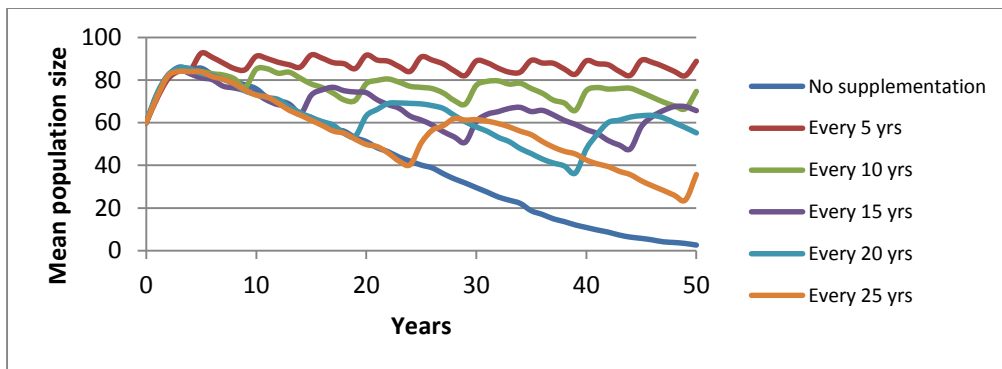


Figure B: Mean population size over time at WORZ, under a supplementation rate of 15 individuals at varied time intervals.

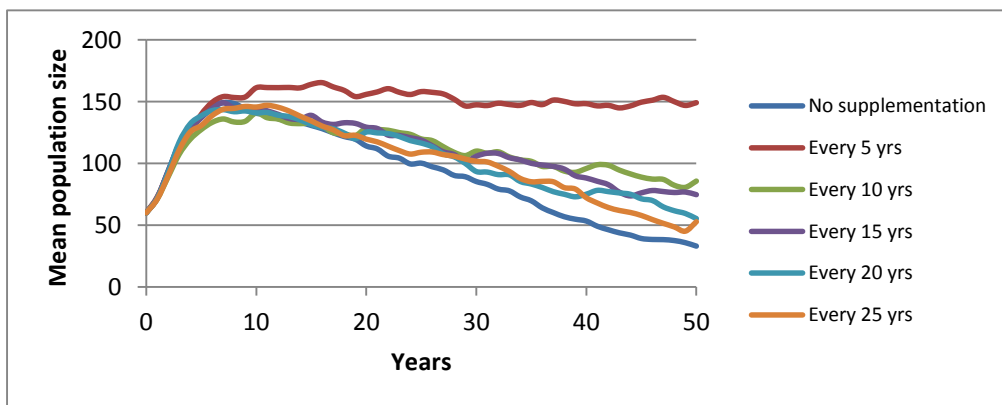


Figure C: Mean population size over time at Mount Rothwell, under a supplementation rate of 8 individuals at varied time intervals.

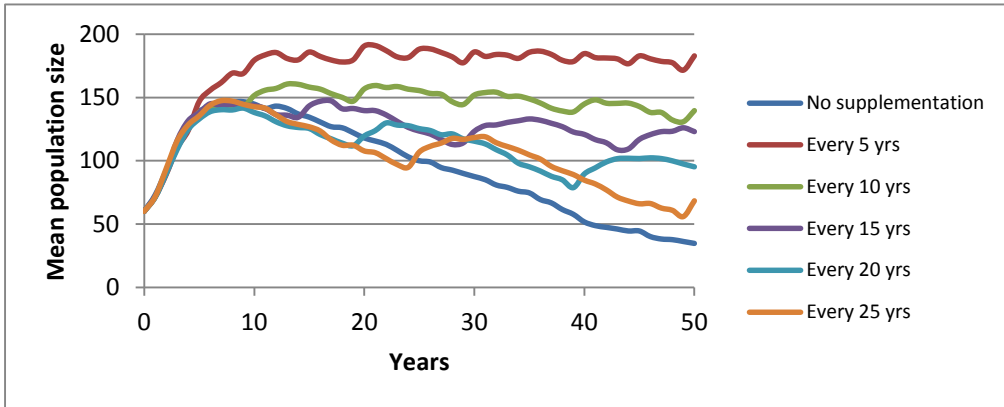


Figure D: Mean population size over time at Mount Rothwell, under a supplementation rate of 15 individuals at varied time intervals.

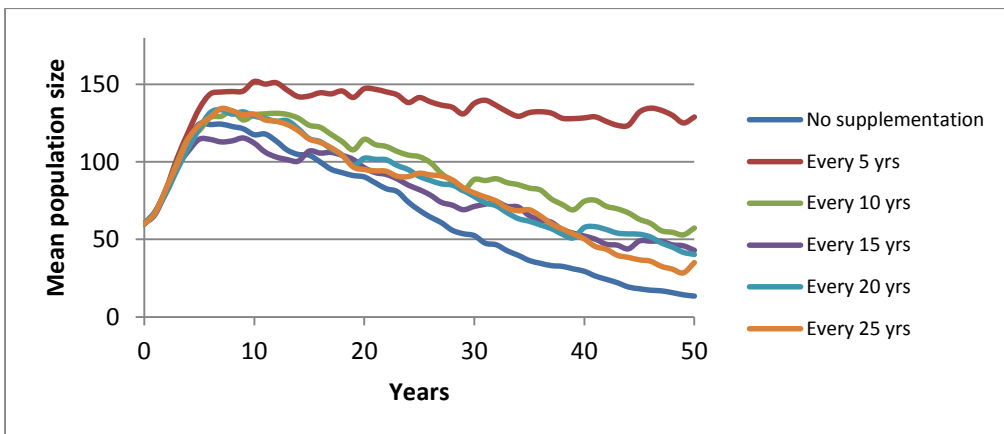


Figure E: Mean population size over time at Woodlands, under a supplementation rate of 8 individuals at varied time intervals.

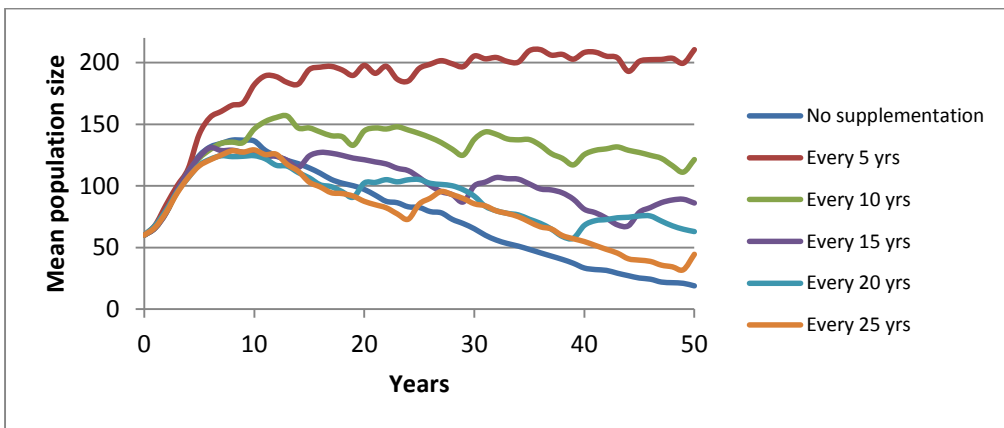


Figure F: Mean population size over time at Woodlands, under a supplementation rate of 15 individuals at varied time intervals.

Table I: Supplementing smaller sites with 8 individuals at different time intervals

Scenario	stoc-r	SD(r)	PE ₅₀	N-all ₅₀	SD(Nall) ₅₀	GeneDiv ₅₀	MeanTE
Hamilton							
At 5yr intervals	0.020	0.259	0	39.63	23.65	0.8405	82.2
At 10yr intervals	0.001	0.257	0	17.34	17.09	0.8577	87.3
At 15yr intervals	-0.003	0.254	0.480	13.61	19.31	0.6899	79.0
At 20yr intervals	-0.008	0.257	0.790	4.85	12.84	0.6187	75.9
At 25yr intervals	-0.011	0.255	0	8.30	3.35	0.8861	73.9
No supplementation	-0.009	0.248	1	0	0	0	70.4
WORZ							
At 5 yr intervals	0.029	0.247	0	73.61	32.62	0.8314	107.6
At 10yr intervals	0.014	0.246	0	48.92	38.05	0.8153	104.9
At 15yr intervals	0.010	0.246	0.210	37.58	35.85	0.7224	113.9
At 20yr intervals	0.005	0.246	0.522	23.20	34.00	0.6521	110.3
At 25yr intervals	0.004	0.245	0	18.84	23.74	0.8486	113.7
No supplementation	0.000	0.239	0.904	2.84	11.18	0.4741	116.5
Mount Rothwell							
At 5 yr intervals	0.021	0.258	0	149.24	86.04	0.8382	99.2
At 10yr intervals	0.006	0.260	0	85.77	90.91	0.8325	93.2
At 15yr intervals	0.006	0.256	0.250	74.70	87.38	0.7437	96.0
At 20yr intervals	0.003	0.256	0.476	55.50	78.29	0.7137	103.8
At 25yr intervals	0.002	0.254	0	52.90	78.09	0.8376	105.4
No supplementation	0.001	0.251	0.712	33.22	70.07	0.6793	106.4
Woodlands							
At 5 yr intervals	0.004	0.270	0	128.89	122.34	0.8464	91.5
At 10yr intervals	-0.011	0.272	0	57.40	94.75	0.8535	89.6
At 15yr intervals	-0.015	0.270	0.448	42.98	87.08	0.7177	82.1
At 20yr intervals	-0.013	0.262	0.660	40.24	86.85	0.7413	87.7
At 25yr intervals	-0.013	0.260	0	35.17	70.06	0.8660	89.5
No supplementation	-0.017	0.260	0.874	13.47	54.21	0.6699	88.0

Table II: Supplementing smaller sites with 15 individuals at different time intervals

Scenario	stoc-r	SD(r)	PE ₅₀	N-all ₅₀	SD(Nall) ₅₀	GeneDiv ₅₀	MeanTE
Hamilton							
At 5 yr intervals	0.038	0.259	0	53.68	20.29	0.8791	81.0
At 10yr intervals	0.015	0.258	0	37.60	22.36	0.8728	78.6
At 15yr intervals	0.008	0.256	0.15	35.37	24.90	0.7876	77.5
At 20yr intervals	0	0.254	0.45	19.33	23.61	0.7062	74.9
At 25yr intervals	-0.006	0.250	0	16.72	7.68	0.9088	75.3
No supplementation	-0.010	0.247	1	0	0	0	70.6

Scenario	stoc-r	SD(r)	PE ₅₀	N-all ₅₀	SD(Nall) ₅₀	GeneDiv ₅₀	MeanTE
WORZ							
At 5 yr intervals	0.043	0.249	0	89.68	25.74	0.8729	106.6
At 10yr intervals	0.026	0.248	0	72.54	35.27	0.8420	105.6
At 15yr intervals	0.019	0.244	0.04	67.77	35.38	0.7924	101.9
At 20yr intervals	0.014	0.247	0.19	54.57	38.81	0.7363	105.1
At 25yr intervals	0.009	0.244	0	34.47	29.51	0.8633	111.9
No supplementation	-0.002	0.247	0.93	1.92	9.37	0.4696	111.6
Mount Rothwell							
At 5 yr intervals	0.030	0.258	0	182.90	76.39	0.8751	87.3
At 10yr intervals	0.017	0.259	0	139.60	90.20	0.8475	88.5
At 15yr intervals	0.013	0.257	0.05	123.00	88.16	0.7993	83.3
At 20yr intervals	0.006	0.259	0.25	95.22	95.32	0.7601	92.2
At 25yr intervals	0.004	0.261	0	68.36	80.86	0.8628	102.8
No supplementation	0	0.252	0.70	34.72	72.55	0.6534	109.6
Woodlands							
At 5 yr intervals	0.017	0.267	0	210.50	115.8	0.8753	79.1
At 10yr intervals	0	0.270	0	121.40	116.84	0.8600	88.7
At 15yr intervals	-0.008	0.272	0.15	86.18	105.17	0.7747	83.0
At 20yr intervals	-0.012	0.267	0.46	62.91	101.94	0.7770	81.9
At 25yr intervals	-0.015	0.267	0	44.50	73.63	0.8907	85.5
No supplementation	-0.014	0.257	0.85	18.88	61.50	0.6654	91.6

Note that probabilities of extinction may be counter-intuitive in some cases. For example PE for 25 year supplementation scenarios may be lower than that for more frequent supplementation. This relates to the year of last supplementation and its distance from year 50. So, for example, supplementing at 25 years will place the final year of supplementation at year 50, so that “empty” sites that would otherwise have been counted as “Extinct” are counted as “Extant”, thereby over-stating the viability of the scenario.