

***DELMA IMPAR* ~ Striped Legless Lizard**

Population & Habitat Viability Assessment





**POPULATION AND HABITAT VIABILITY ASSESSMENT (PHVA)
FOR THE**

STRIPED LEGLESS LIZARD

(Delma impar)

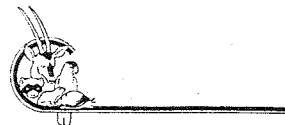
Based on a workshop held in Canberra, Australia
30 July - 2 August 1996


FINAL REPORT

Prepared by Workshop Participants

A Collaborative Workshop:

Australian Capital Territory Parks and Conservation
Striped Legless Lizard Working Group
Australasian Regional Association of Zoological Parks and Aquaria (ARAZPA)
Conservation Breeding Specialist Group (SSC/IUCN)



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DELMA IMPAR
Striped Legless Lizard

Population & Habitat Viability Assessment Workshop

Canberra

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Section 1 ~ **Executive Summary**



Executive Summary

OVERVIEW

The Striped Legless Lizard *Delma impar* (Fischer 1882) is a reptile of the family *Pygopodidae*. The average snout-vent length of adults is 90 mm (Cogger 1992), with a maximum total length of about 300 mm and an average body weight of 4.1 grams (Coulson 1990). Sexes are externally similar.

Legless lizards superficially resemble small snakes. However, they can be readily distinguished from snakes by having a visible ear opening, fleshy broad tongue, the presence of remnant hindlimbs (which are reduced to scaly flaps near the vent) and a tail that is longer than the body, which may be voluntarily shed.

The Striped Legless Lizard is found primarily in lowland native grasslands (Coulson 1990, Osborne *et al.* 1993). This habitat type occurs on flat or gently undulating plains (Coulson 1990, Hadden 1995), and is dominated by perennial, tussock-forming grasses such as kangaroo grass *Themeda triandra*, speargrass *Stipa* spp. and wallaby grass *Danthonia* spp. The species is also found in some areas dominated by exotic grasses (Coulson 1990, Williams and Kukolic 1991, Kukolic *et al.* 1994, Rauhala *et al.* 1995, Hadden 1995). A tussock structure in grassland appears to be an important habitat characteristic (Wildlife Research Unit 1992, Hadden 1995), although little is known about the way in which the vegetation is utilised. Some evidence exists to suggest that lizards over-winter at the base of grass tussocks or just below the soil surface (Wildlife Research Unit 1994).

The Striped Legless Lizard is confined to south-eastern Australia. It is currently known to occur at several localities in Victoria, mainly on the basalt plains to the north and west of Melbourne and in the western district of the state (Department of Conservation and Environment 1992). In NSW, the only recent records of the species are from Batlow (1977), Goulburn (1992) and Cooma (1995) (Cogger 1994, Husband 1995, R. Pietsch *pers. comm.*). *Delma impar* has also been recorded from South Australia, in the extreme south-eastern corner of the state, however the most recent records from this area are in 1969, and the area now appears unlikely to support a population of the species (Coulson 1990, Hadden 1995). In the ACT, *Delma impar* is found in four disjunct localities; Gungahlin, the Majura Valley, Yarramundi Reach and the Jerrabomberra Valley (Rauhala *et al.* 1995).

Throughout its range in south-eastern Australia, the Striped Legless Lizard is considered to have suffered a substantial contraction in its distribution since European settlement. An investigation by Coulson (1990) indicated that there were few recent records of the species from areas of western Victoria where it had previously been recorded. Subsequent work by Hadden (1995) estimated that of the 125 sites from which the species has been recorded historically, it is now likely to occur in as few as 40 of those sites.

It is believed that a combination of factors, including clearing of grasslands for urban development, some agricultural practices (prolonged heavy grazing by stock, pasture improvement, crop production), habitat fragmentation, weed invasion and inappropriate fire regimes have contributed to the decline of the species (Cogger *et al.* 1993).

D. impar is recognised as a threatened species at international, national, and local levels:

International

- Vulnerable (IUCN 1994)

National

- Vulnerable (ANZECC 1991)
- Vulnerable (Sched 1, Part 2 of the *Endangered Species Protection Act* 1992 [Commonwealth])

Local

- Vulnerable in the Australian Capital Territory (*Nature Conservation Act 1980* [ACT])
- Vulnerable & rare species in New South Wales (Sched 12, *National Parks & Wildlife Act 1974* [NSW])
- Threatened taxon in Victoria (Sched 2, *Flora & Fauna Guarantee Act 1988* [Vic])
- Vulnerable in Victoria (CNR, 1995)

Twenty-four biologists, managers, and policy makers attended a Population and Habitat Viability Assessment (PHVA) Workshop in Canberra, Australia at the ACT Parks & Conservation Service O'Connor offices on July 30 - August 2 1996 to apply the recently developed procedures for risk characterization and assessment and formulation and testing of risk management scenarios to the Striped Legless Lizard (*Delma impar*). The workshop was proposed by the Australian Capital Territory Parks and Conservation Service's Wildlife Research Unit and was a collaborative effort of the Australasian Regional Association of Zoological Parks and Aquaria (ARAZPA), the Striped Legless Lizard Recovery Team and the Conservation Breeding Specialist Group (CBSG) of the Species Survival Commission / World Conservation Union (SSC/IUCN). The purpose was to review data from the wild populations and captive animals as a basis for assessing extinction risks, assessing different management scenarios, and developing stochastic population simulation models. These models estimate risk of extinction and rates of genetic loss from the interactions of demographic, genetic, and environmental factors as a tool for ongoing management of the subspecies. Other goals included determination of habitat and capacity requirements, determination of species distribution and numbers, and prioritized research needs.

The first day consisted of a series of presentations summarizing data from the wild populations in NSW, Victoria, and ACT, and broad management strategies to date. After presentations on small population biology, the PHVA process, and an interactive VORTEX exercise, the participants formed three working groups (wild population habitat, wild populations, and population biology and modeling) to review in detail current information, to hear all ideas, and to develop management scenarios and recommendations. Stochastic population simulation models were developed and initialized with ranges of values for the key variables to estimate the viability of the wild population using the VORTEX software modeling package. Using data compiled from the literature and by consultation with workshop participants, a series of agreed estimates of baseline population values for the parameters required by the Vortex program were developed. These were then used to model specific populations across the range of the species, characterized in terms of urban or rural environment, size, presence of grazing, effects of fires, and predation.

This workshop report includes a set of recommendations for research and management of the wild habitat and wild populations, as well as sections on the life history of the species (as far as is currently known), distribution of the known populations and the population biology and simulation modeling of the population.

RECOMMENDATIONS

Research

1. Develop improved and reliable detection and external sexing techniques (including capture, marking and tracking). This is critical for accurate data analysis and population assessment.
2. Investigate key life history parameters including longevity, behaviour, movement, age-specific mortalities, and female reproductive success; with age-specific mortality rates perhaps the most important to clarify.
3. Determine types and rates of predation on both urban and rural *D. impar* populations, and investigate appropriate predator control techniques, particularly for urban *D. impar* populations.
4. Investigate the impacts of fire frequency and intensity on *D. impar* populations.

5. Undertake a budgetary, resource and priority analysis of management and research requirements, and target possible funding sources and potential participants. This should be co-ordinated by the Striped Legless Lizard National Recovery Team.
6. Investigate the carrying capacity of different habitat sizes and qualities and the biological and climatic processes which may impact on carrying capacity.
7. Model viability of clusters via PHVA analysis.
8. The Recovery Team should develop a GIS database of all sites/populations, cluster boundaries, habitat variables and genetic information. Establish monitoring programs at key sites to assess effectiveness of conservation management strategies, population sizes and structures to determine if any supplementation or movement of animals is required between populations.
9. Develop criteria for, and undertake a trial translocation, including monitoring survival and impact on source populations.

Management

10. Adopt, refine and implement the cluster approach.
11. Prepare management strategies for urban areas, particularly the Gungahlin (ACT) reserves as a priority, as these areas contain the most significant known populations of *D. impar*. Management issues to include: grazing management; fire fuel management; weed control; fencing; management and recreation access; monitoring; predator control; neighbour relations; priority setting. Mechanisms to ensure proper documentation of all activities and to resolve conflicting management objectives (e.g. between promoting floral diversity and protecting *D. impar* populations) need to be included.
12. Ensure all stakeholders have opportunities for involvement in management programs for *D. impar* populations and are kept informed of new information. The Recovery Team could co-ordinate a monitoring protocol to promote consistency across all management areas.
13. Sites that are essential for the viability of *D. impar* clusters should be acquired or land-owners approached regarding voluntary conservation agreements.
14. Develop materials that raise community awareness and understanding of the importance of native grasslands as habitat for *D. impar* and the possible role and impact of domestic cats or other carnivores as predators.

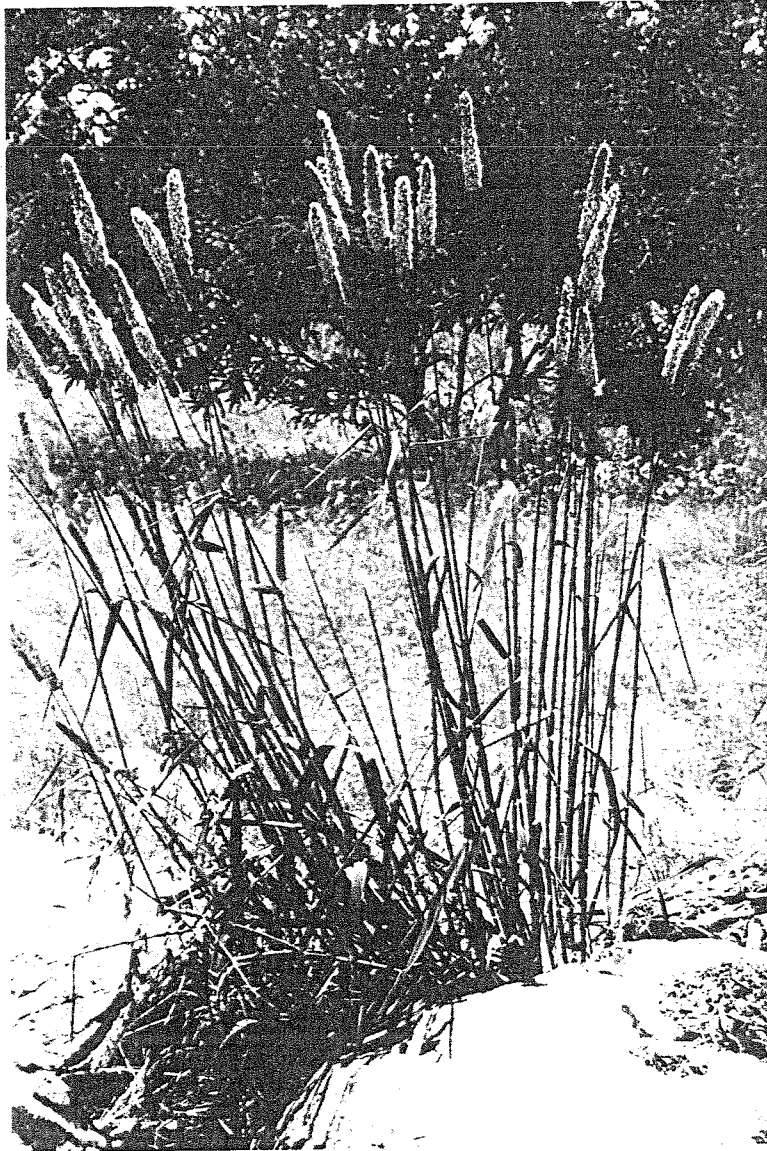
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Section 2 ~ Overview & Participants

Invitation

Population & Habitat Viability Analysis for the Striped Legless Lizard (*Delma impar*)

ACT Parks and Conservation Service
Canberra, Australia
30 July - 2 August, 1996

This notice confirms arrangements for the Population and Habitat Assessment workshop on *Delma impar*.

Dr Ulysses Seal (Chair of the IUCN/Species Survival Commission Captive Breeding Specialist Group) and Dr Phil Miller will be conducting the Workshop. The workshop is being organised with the assistance of Mr Jonathan Wilcken, Acting Conservation Coordinator, Australasian Regional Association of Zoological Parks and Aquaria.

The Workshop is arranged in association with the National Recovery Team for the Striped Legless Lizard, the NSW National Parks and Wildlife Service (Southern Zone) and the University of Canberra.

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- Dates:** Commencing 9.00 a.m., Tuesday 30 July to Friday 2 August 1996 (inclusive). Field visits will be arranged on Monday 29 July for those who are unfamiliar with the ACT's grassland reserves and the ACT distribution of *Delma impar*.
- Venue:** ACT Parks and Conservation Service, cnr Dryandra Street and Belconnen Way, O'Connor, ACT 2602.
- Accommodation:** Convenient accommodation is available at the Canberra Motor Village, only five minutes walk from the Workshop venue. Bookings (toll free) 1800 026 199.
- Cost:** Participants are asked to make their own accommodation arrangements. Lunches will be organised at the workshop venue with the cost being divided between participants. Evening meals will be organised each day depending on individual and group initiatives and the cuisine desired.
- Enquiries to:** David Shorthouse
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Overview

1. Species Description and Distribution

Prepared from unpublished material compiled by Wildlife Research Unit, ACT Parks & Conservation Service, and the Victorian Striped Legless Lizard Working Group.

DESCRIPTION

The Striped Legless Lizard *Delma impar* (Fischer 1882) is a reptile of the family Pygopodidae. The average snout-vent length of adults is 90 mm (Cogger 1992), with a maximum total length of about 300 mm and an average body weight of 4.1 grams (Coulson 1990). Sexes are externally similar.

The species is variable in colour but is most commonly grey-brown above, with a series of dark brown or blackish longitudinal stripes along the length of the body and tail, commencing at the neck (Cogger 1992). A large amount of variation exists between individuals in colour and intensity of the striping, and in some animals (particularly in the young), striping is indistinct or absent. The colour of the head is darker than that of the body, being dark brown to dark slate grey in adults and black in young individuals. The ventral surface has been described as whitish (Cogger 1992), however some individuals have salmon-pink coloration on the flanks that may extend to the undersurface. Most individuals have yellow coloration on the infralabial and adjacent gular scales, extending back to the tympanum (Coulson 1990).

The Striped Legless Lizard can usually be distinguished, by the presence of stripes, from the Olive Legless Lizard *Delma inornata*, a closely related species which also occurs in Victoria, NSW, and the ACT region.

Legless lizards superficially resemble small snakes. However, they can be readily distinguished from snakes by having a visible ear opening, fleshy broad tongue, the presence of remnant hindlimbs (which are reduced to two scaly flaps near the vent) and a tail that is longer than the body, and which can be voluntarily shed.

HABITAT

The Striped Legless Lizard is found primarily in lowland native grasslands (Coulson 1990, Osborne *et al.* 1993). This habitat type occurs on flat or gently undulating plains (Coulson 1990, Hadden 1995), and is dominated by perennial, tussock-forming grasses such as kangaroo grass *Themeda triandra*, speargrass *Stipa* spp. and wallaby grass *Danthonia* spp. plains (Coulson 1990, Hadden 1995). The species is also found in some areas dominated by exotic grasses (Coulson 1990, Williams and Kukolic 1991, Kukolic *et al.* 1994, Rauhala *et al.* 1995, Hadden 1995). A tussock structure in grassland appears to be an important habitat characteristic (Wildlife Research Unit 1992, Hadden 1995), although little is known about the way in which the vegetation is utilised. Some evidence exists to suggest that lizards over-winter at the base of grass tussocks or below the soil surface (Wildlife Research Unit 1994).

A recent review of sites in Victoria and the ACT, where the species is known to occur (Hadden 1995), has determined that habitat is characterised by a vegetative cover dominated by native tussock grasses, and that soils generally have a moderate to high clay content which often produce cracks in summer. In Victoria, most sites supporting the species have a cover of lightly embedded rocks, although this is not a feature of the species' habitat in the ACT.

Although the Striped Legless Lizard is found in both primary and secondary grasslands, Dorrrough (1995) found it to inhabit secondary grasslands only if these were within 2km of primary grasslands.

Most areas where the species persists are thought to have had low to moderate levels of agricultural disturbance in the past (Coulson 1990, Hadden 1995, Dorrrough 1995). It has been suggested (Coulson 1990, Dorrrough 1995) that ploughing may be a practice that is particularly incompatible with the survival of the species in an area.

BEHAVIOUR AND BIOLOGY

The Striped Legless Lizard is known to feed on a variety of insects and arthropods, including spiders, crickets, cockroaches and caterpillars (Coulson 1990, Wainer 1992, Nunan 1995). Some evidence is available to suggest that the species displays some selectivity in its diet, with *Lepidoptera* larvae (caterpillars) being implicated as a particularly important food resource (Nunan 1995).

The species is diurnal and surface active from late spring to early autumn, with a peak in activity in November and December (Kukolic 1994). Gravid individuals are commonly caught in these months, with two eggs being laid in December. Some evidence is available to suggest that communal oviposition occurs and that at least sometimes, eggs may be laid under rocks or other substrate (Mills 1992, Rauhala 1996). Incubation periods of between 35 and 60 days have been observed in captivity, under ideal conditions. However, the incubation period is likely to be longer in the field.

The longevity of the species is not known, but a maximum of ten years has been estimated (Webster *et al.* 1991, Dorrough 1995).

DISTRIBUTION

The total geographic range of the species is confined to south-eastern Australia. It is currently known to occur at several localities in Victoria, mainly on the basalt plains to the north and west of Melbourne, and in the western district of the state (Department of Conservation and Environment 1992). In NSW, the most recent isolated records are from Cooma in 1995 (BHP/Westcoast Energy 1995), Goulburn in 1992 (Husband 1995) and Batlow in 1977 (Cogger *et al.* 1993). The species has also been recorded from South Australia in the extreme south-eastern corner of the state; however, the most recent records from this area are in 1969 and the area now appears unlikely to support a population of the species (Coulson 1990, Hadden 1995).

In the ACT, the potential range of the species prior to European settlement is likely to have been within the more or less continuous area of treeless plains which extended over 20,000 hectares. However, most of this area has been developed for urban and related purposes, and the current distribution of the Striped Legless Lizard in the ACT is a fragmented one, with four disjunct populations recognised; ie. Gungahlin, Yarramundi Reach, Majura Valley and the Jerrabomberra Valley (Rauhala *et al.* 1995). The evidence available to date suggests that areas of Gungahlin are the stronghold for the species in the ACT region, although nearby areas of New South Wales and areas of potentially suitable habitat beyond the ACT border are yet to be thoroughly investigated.

Throughout its range in south-eastern Australia, the Striped Legless Lizard is considered to have suffered a substantial contraction in its distribution since European settlement. An investigation by Coulson (1990) indicated that there were few recent records of the species from areas of western Victoria where it had previously been recorded. Subsequent work by Hadden (1995) estimated that of the 125 sites from which the species has been recorded historically, it now occurs in as few as 40 of those sites.

It is believed that a combination of factors, including clearing of grasslands for urban development, some agricultural practices (prolonged heavy grazing by stock, pasture improvement and crop production), habitat fragmentation, weed invasion and inappropriate fire regimes have contributed to the decline of the species (Cogger *et al.* 1993).

2. National Recovery Team & Working Groups

The management of the Striped Legless Lizard throughout its range is co-ordinated by the Striped Legless Lizard National Recovery Team, which was established in December 1995. The Recovery Team's primary conservation goal is to ensure the long-term survival of *Delma impar* throughout its distribution. Its terms of reference are:

- To establish strategic objectives and research and management priorities for the conservation of the Striped Legless Lizard throughout its distribution.
- To facilitate the co-ordination of the National and State / Territory management strategies and to participate in their review.
- To monitor and evaluate success of research and management programs.
- To assist with seeking resources for conservation programs and to advise on their effective use.
- To provide for exchange of information on the conservation of the Striped Legless Lizard and provide authoritative advice as required.
- To foster community awareness of and the establishment of extension programs for the conservation of the Striped Legless Lizard.
- To integrate Striped Legless Lizard research and management with wider grasslands conservation efforts (and organisations).

The Recovery Team comprises representatives of the Victorian Striped Legless Lizard Working Group (VSLLWG), the ACT / NSW Striped Legless Lizard Working Group, the Australian Nature Conservation Agency (ANCA) and the South Australian Department of Environment and Natural Resources. The two Working Groups maintain close contact and co-ordinate management and research in their respective areas.

The VSLLWG was established in December 1990 and is the body recommended in the Flora and Fauna Guarantee Action Statement for *D. impar* to co-ordinate management actions within Victoria (Webster *et al.*, 1992). The Group is a voluntary and independent organisation, with multi-disciplinary membership. The major activities and tasks can be divided into four main areas, each largely overseen by different agencies, ie. Research (University of Melbourne, Victoria University of Technology / St. Albans), management (Department of Environment and Natural Resources), captive populations and education (Melbourne Zoo), and community awareness and volunteer groups (Victorian National Parks Association) (Kutt *et al.*, 1995). However, there is extensive co-operation and collaboration on programs.

The ACT-NSW Striped Legless Lizard Working Group brings together government, university and community expertise in the conservation and management of *Delma impar* in the ACT and NSW. The objectives of the group are to coordinate surveys for the species in the ACT and Southern Tablelands region of NSW, to establish priorities and coordinate research and conservation management for the species and to encourage community awareness of and participation in the conservation of the species.

3. CBSG & PHVA's

The Captive Breeding Specialist Group (CBSG) of the IUCN / SSC is a global network of volunteer experts from zoos, wildlife agencies, and universities whose mission is "to assist the conservation and establishment of viable populations of threatened species through captive propagation programs and through intensive protection and management of small and fragmented populations in the wild." CBSG serves to catalyze cooperation and link programs between the wildlife and captive conservation communities worldwide.

The PHVA workshop, as developed by CBSG, provides viability assessments for each population of a species or subspecies determined to be deserving of more conservation attention. The assessment undertakes an in-depth analysis of information on the life history, population dynamics, ecology, and population history of species, subspecies and populations. Information on demography, genetics, and environmental factors pertinent to assessing population status and risk of extinction under current management scenarios and perceived threats are assembled in preparation for the PHVA and during the workshop.

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Striped Legless Lizard

Population & Habitat Viability Assessment Workshop

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Section 3 ~ **Habitat Assessments**



Habitat Assessment

Habitat Working Group

Participants:

- David Shorthouse (ACT Parks and Conservation Service),
- Josh Dorrrough (World Wide Fund for Nature)
- Odile Arman (ACT Parks and Conservation Service)
- Peter Mills (ACT Parks and Conservation Service)
- Simon Clark (Conservation Council of South East Region and Canberra)

Conservation goal

Maintain areas of suitable habitat able to support known populations of *Delma impar* throughout the range of its distribution (Overlap with other group)

Objectives of working group

- Define *D. impar* habitat
- Assess threats to *D. impar* habitat
- Identify management guidelines and research needs for *D. impar* habitat

Delma impar habitat requirements

Delma impar is a grassland dependent species, currently found in primary natural temperate grasslands; and in grassy woodlands and secondary grasslands near to primary grassland. Surveys have shown that *D. impar* is also present in modified grasslands, with significant content of exotic grasses, including *Phalaris* and other species providing suitable tussock structure. Research to this date has failed to show that *D. impar* has specific floristic requirements.

The current distribution of *D. impar* may not reflect its historical range. In the ACT and around Melbourne, primary grasslands have been largely developed for urban use, resulting in remaining populations being found in fringing habitat. In addition, some populations may be the result of dispersal into secondary grasslands - these areas derived from grassy woodland ecosystems from which tree cover has been largely or entirely removed, leaving the grassy understory intact.

Throughout the range of *D. impar*, there are known and likely primary and secondary grasslands that no longer contain the species, colonisation being possibly prevented by land use, time and other barriers to dispersal. Secondary and some exotic grasslands provide opportunities for alternative conservation strategies should primary habitat become unsuitable.

Grasslands have historically been exploited by grazing and other agricultural practices, along with urban development and infrastructure. This grassland exploitation, together with changed fire regimes, rabbits and droughts, may have caused a major decline of the species, resulting in the fragmented distribution seen today. Other contributory factors may have been fox and cat predation.

Key habitat features appear to be:

1. Presence of cover in the form of:
 - continuous vegetative cover
 - rocks (Victoria)
 - tussock structure to grasses
 - cracking soils (Victoria)
 - animal burrows

2. Presence of a diversity and abundance of arthropod fauna:
 - e.g. spiders, crickets, cockroaches. The relationship between arthropod fauna and grassland as habitat needs to be determined in order to assist understanding of the food requirements of *D. impar*.
3. Moisture requirement

Threats

The working group compiled the following Table 3.1 by assessing likely impacts of various threats to *D. impar* populations in both urban and rural situations, as identified in the ACT. The objective was to provide the modelling group with estimates of how threats or catastrophes might affect mortality rates and carrying capacities:

Table 3.1- Likely impacts of various threats to *Delma impar* in urban and rural situations

URBAN HABITAT	RURAL HABITAT
<p>WILDFIRE, including arson-lit fire</p> <p>Have both habitat and population effects</p> <p>Season Autumn/winter - low impact Spring /summer - high impact Manage edges to mitigate impact</p>	<p>WILDFIRE</p> <p>Summer fires Similar effects, response time lower, fuel loads lower ? except where <i>D. impar</i> is? (e.g. Travelling Stock Route)</p>
<p>MANAGEMENT FIRES</p> <p>Compartmentalise reserve Habitat management - <i>D. impar</i> impact? Rocks may provide fire refuges (scope for artificial rocks?) Different types of management fires have different effects on habitat and <i>D. impar</i></p>	<p>MANAGEMENT FIRES</p> <p>Low threat to <i>D. impar</i> , habitat</p>
<p>DROUGHT</p> <p>Low impact on habitat Duration effects on <i>D. impar</i> population</p>	<p>DROUGHT</p> <p>High impact if grazed Possible greater impact on young/breeding <i>D. impar</i> Duration effects on <i>D. impar</i> population could be high if area grazed and drought continues</p>

URBAN HABITAT (cont.)**RURAL HABITAT (cont.)**

GRAZING	GRAZING
Managed grazing for conservation may have low impact and possible positive conservation effects. May be significant fire control mechanism	Low intrinsic impact but may be compounded by other threats such as fire and drought Impact dependent on other suitable & available cover Need co-operation of landholder
FERAL ANIMALS	FERAL ANIMALS
Impact undefined (research question); likely to be more critical in isolated and small populations in urban areas rather than rural areas Urban cat and dog policy Fox control Rabbit/RCD/warren ripping Mice populations and prey switching by predator	Impact undefined (research question) Feral cat and fox control
NATIVE PREDATORS	NATIVE PREDATORS
Unknown impact, but likely to be greater than rural. Brown snakes, Blue-tongues increased numbers in urban open space Perches for predatory birds	Low impact
LAND MANAGEMENT PRACTICES	LAND MANAGEMENT PRACTICES
Urban run-off Herbicides Fertiliser Salinity Disturbance effects by other agencies (e.g. groundwater, pipelines, etc.)	Herbicides Fertiliser Cultivation Salinity Disturbance effects by other agencies (e.g. groundwater, pipelines, etc.)
IGNORANCE	IGNORANCE
Status and reason for reserved areas Operation and impact of all above threats Involvement of local community	Operation and impact of all above threats Involvement of land owner and local community

Weeds are another threat which is not included in the above table.

The working group then repeated the exercise, but considered combinations of threats and assessed how the interaction of threats may affect mortality and carrying capacity (see later Table). In particular, drought and grazing are likely to combine in the total effect they have on the carrying capacity of *D. impar* habitat.

Predator control in urban areas must be addressed and constraints on control measures reviewed, in particular foxes and cats.

The effects of salinity are unknown.

Having identified the range of threats and likely impact on *D. impar* populations and habitat, the group then assessed each threat in terms of its probability of occurring and its likely impact if it did occur. We used information from the Population Working Group to establish four scenarios, sites smaller than 20ha and larger than 20 ha, in both urban and rural locations.

Table 3.2 . Identification of threats & assessment of their possible impacts on urban & rural *D. impar* areas.

	URBAN <20ha Prob/impact	URBAN >20ha Prob/impact	RURAL <20ha Prob/impact	RURAL >20ha Prob/impact
PREDATORS				
managed sites	H/H	H/M	H/L	H/L
unmanaged sites	H/H	H/M	H/L	H/L
GRAZING				
managed sites	M/L	M/L	H/L	H/L
unmanaged sites	H/L	H/L	H/L	H/L
FIRE				
managed sites	H/H	H/M	L/L	L/L
unmanaged sites	H/H	H/M	L/H	L/H
DROUGHT				
managed sites	H/L	H/L	H/L	H/L
unmanaged sites	H/L	H/L	H/L	H/L
WEEDS				
managed sites	L/L	L/L	M/L	M/L
unmanaged sites	M/L	M/L	M/L	M/L
CULTIVATION				
managed sites	n/a	n/a	L/H	L/H
unmanaged sites	L/H	L/H	M/H	M/H
SOIL DISTURBANCE				
managed sites	L/L	L/L	L/L	L/L
unmanaged sites	H/L	H/L	H/L	H/L
FERTILISER				
managed sites	n/a	n/a	L/L	L/L
unmanaged sites	L/L	L/L	H/M	H/M
SALINITY				
managed sites	?	?	?	?
unmanaged sites	?	?	?	?
HERBICIDES				
managed sites	H/L	H/L	H/L	H/L
unmanaged sites	L/M	L/M	L/M	L/M
RECREATION				
managed sites	H/M	H/L	L/L	L/L
unmanaged sites	M/H	M/H	L/L	L/L

Definition of terms (table 3.2)

Managed - land managed for conservation purposes either by government agency or through conservation agreements with private landholders.

Unmanaged - land not managed for conservation purposes and including rural leases and freehold.

Probability of occurrence

High (H), Medium (M) and Low (L)

Impact on habitat if threat does occur

High (H), Medium (M) and Low (L)

The Table 3.2 identified the relationship between a range of threats and the urban and rural managed and unmanaged areas. The threats can act singly or in combination with each other. In combination they may compound or mitigate the effect on *D. impar* populations.

Table 3.3 summarises the interaction of major threats and their likely impacts on *D. impar*.

Table 3.3. Interaction of major threats in urban and rural *D. impar* areas.

COMBINED THREATS	URBAN	URBAN	RURAL	RURAL
	<20ha Prob/impact	>20ha Prob/impact	<20ha Prob/impact	>20ha Prob/impact
FIRE & DROUGHT				
Managed	H/H	H/H	H/H	H/H
Unmanaged	H/H	H/H	H/H	H/H
GRAZING then FIRE				
Managed	M/M	M/M	L/M	L/M
Unmanaged	L/M	L/M	H/M	H/M
FIRE then GRAZING				
Managed	n/a	n/a	n/a	n/a
Unmanaged	M/H	M/H	H/H	H/H
RECREATION then FIRE				
Managed	H/M	H/M	L/L	L/L
Unmanaged	H/M	H/M	L/H	L/H

Probability of occurrence

High (H), Medium (M) and Low (L)

Impact on habitat if threat does occur

High (H), Medium (M) and Low (L)

Predicted Impacts of Specific Threats on Populations of *Delma impar*. Information to assist with population modelling.

Drought

Severe drought (catastrophic) appears to occur in the A.C.T. on average once in every 25 years. Impact on *D. impar* will be largely dependent on whether the site is grazed or ungrazed. Loss of vegetative cover, decline in invertebrate prey and lack of moisture will have the greatest influence on *Delma impar* population. Adults may be able to survive during these periods and for this reason the impact is believed to largely effect recruitment and juvenile survivorship. As indicated in the above table, the effects of grazing and drought will be interlinked, in grazed areas recruitment may be as low as 0. Trapping in the A.C.T. appears to support this view, with low captures during a drought year and low numbers of juveniles caught in the subsequent year. Of particular interest is the result that capture rates of adults may return to pre-drought levels in subsequent years.

Grazing

Grazing results in a loss of vegetative cover, soil disturbance and weed invasion. Changes in biomass and vegetative cover are believed to have the greatest influence of populations of *D. impar*. Grazing is known to alter the abundance, diversity and distribution of invertebrates and may in turn influence the distribution and

abundance of *D. impar*. The effect of grazing alone will be extremely variable. However, it is most likely to influence carrying capacity, recruitment and survivorship (increased predation).

Fire

Based on available data, fire frequencies are believed to vary significantly. Furthermore, a range of management or control fire regimes is possible. The impact of these fires will vary depending on location (urban, rural, state), timing and seasonality. While control burns may cover a smaller area, they remove a larger amount of vegetative cover and are believed to be more intense (see Table 3.4). This is perhaps of greater significance at sites where cover such as rocks or cracks in the soil are available. Fires are believed to cause some direct mortality and an increase in predation may also be associated with these events. Loss of habitat and a reduction in invertebrate prey will result in a decline in recruitment success.

Table 3.4 Different types of fires and their effects on *Delma impar*

	Wildfire	Management Fire	
		Control	Convection
Ground level temperature	235-350 °C	350-458 °C	350-458 °C
Fuel removed	40%	75%	90%
Frequency	rural 1:25 yrs urban 1:1 year	1 year / 3 yrs / 10yrs potential frequencies	Victoria
			2yrs / 3yrs/ 10yrs potential frequencies
% area burnt	rural 80%, urban 10%	urban 20%; rural 80%	25%
Season	Summer / Feb / April	urban Sept / Oct rural March / May	March / May
Microhabitat temp. (Rocks, cracks)	very low	very low	very low
<i>D. impar</i> mortality	lower; over large rural area or small urban area	higher; over large rural area small urban area	

Predation

Some data appears to suggest that predation could be a strong influencing factor on population size and may result in age-specific mortality. Data from tail-loss records indicate that some 30-40% of *D. impar* caught may have been attacked by a predator. However, these data do not provide any indication of the proportion of animals actually taken by a predator. The data also indicate that tail-loss may be higher in urban sites than rural sites. Some preliminary evidence from urban areas indicates that juvenile mortality may be higher than adult mortality.

Management of threats and the need for research to inform management

A range of threatening processes have been identified. However, very little is known of their separate and combined effects on *D. impar* populations. Managers often have skills in managing individual threats, but may not have the experience in managing a combination of threats, particularly in relation to a threatened species such as *D. impar*.

Management of threats should include the possibility of establishing learning situations that inform the manager

and raise questions that can be the subject of specific research projects. Sharing of experience and information between managers responsible for *D. impar* populations can be a valuable mechanism.

It may be necessary to establish some research trial plots to monitor the effects of separate and combined threatening processes. These plots may have to be located on the edge or buffer zones of the management area. Research to answer managers' questions is the top priority.

Research topics include fire as a mortality factor on *D. impar*; the relationship between fire and structure of grassland; recovery of *D. impar* after fire; *D. impar* recolonisation of burnt areas; and the role of refuges in fire. Similar questions can be posed for other threats including drought, grazing and weeds.

There is an urgent need for research on predators and their effect on *D. impar* populations, including :

- impact of predation in *D. impar* populations
- predator definition, ie. Cats, foxes, raptors, etc.
- proximity to urban areas
- age-specific predation mortality in *D. impar*
- management actions to instigate predation effects
- relevance of urban structures to predation, eg. Poles, signs, and street trees

Management of predators, particularly in urban areas can present some special difficulties. For example, there are constraints on available techniques such as poisoning foxes and control of domestic cats, which may need to be reviewed if safety issues can be resolved. Education of neighbouring residents may assist with control of domestic cats. Cat-free zones and cat curfews may be alternatives for investigation.

Very little is known of the effects of recreation on *D. impar* populations and habitat in urban areas. Research results indicate that *D. impar* do not cross bare ground and, therefore, permanent trails (both for fire management and recreation) are likely to fragment a conservation area.

It is not known what effect herbicides used for weed control in *D. impar* areas may have on *D. impar*. Control methods may also be an important issue, noting that, spot spraying of weeds is the preferred method at this time. In rural areas land holders must be provided with information about the latest products and methods for weed control in *D. impar*- sensitive areas.

Habitat Management Working Group Summary of Recommendations

Research

1. Seek co-operation of university and other organisations to establish and conduct research programs. This should be co-ordinated by the National Recovery Team.
2. Encourage research into the impact of predation on the structure and growth of *D. impar* populations; and site-specific levels of predation (see text for specific research questions)
3. Investigate alternative control techniques for the management of predators.
4. Investigate the impact of fire frequency and intensity on *D. impar* populations.
5. Assess the role of varying climatic conditions, including periods of low and very high rainfall, on the survival, recruitment and carrying capacity of habitat for *D. impar* populations.
6. Assess the effectiveness and impact of alternative management techniques, other than fire, for reducing biomass, e.g. grazing and slashing.

7. Investigate the food requirements of *D. impar*, particularly the relationship between arthropod fauna and grassland habitat.

Management

8. Undertake a budgetary, resource and priority analysis of management and research requirements and target possible funding sources. This should be co-ordinated by the National Recovery Team.
9. Prepare management strategy/plans for urban areas, particularly the Gungahlin reserves as a priority, as these areas contain the most significant known populations of *D. impar*. Management issues to include: grazing management; fire fuel management; fencing; management and recreation access; monitoring; predator control; neighbour relations; and priority setting.
10. Management plans for *D. impar* need to include mechanisms to resolve conflicting management objectives, such as promoting grassland species diversity through implementing control burning programs which may be detrimental to *D. impar* populations.
11. Involve field staff and the community in monitoring programs for *D. impar* populations. The Recovery Team could co-ordinate a monitoring protocol to promote consistency across all management areas.
12. Ensure field managers document management activities and threat control programs in *D. impar* areas.
13. Determine the most appropriate methods of weed control within *D. impar* conservation areas. Assess the impact of other management techniques on weed invasion.

Community awareness and education

14. There is a need to ensure close liaison between all stakeholders and conservation agencies to ensure that appropriate land management practices are implemented. This should be co-ordinated by the National Recovery Team.
15. Encourage all stakeholders to participate in workshops, field visits and relevant conferences.
16. Sites that are essential for the viability of *D. impar* clusters should be acquired or land-owners approached regarding voluntary conservation agreements.
17. Raise community awareness and understanding of the importance of native grasslands as habitat for *D. impar* through print and electronic media, interpretative walks, school visits, community forums, brochures and signs.
18. Develop educational materials that raise community awareness of the (possible) impact of domestic cats on *D. impar* and what house-holders can do to minimise that impact.

DELMA IMPAR
Striped Legless Lizard

Population & Habitat Viability Assessment Workshop

Canberra

30 July - 2 August 1996

FINAL REPORT



Section 4 ~ **Population Assessment**



Populations *Delma impar* Group

Participants

- Peter Robertson (Dept. Natural Resources & Energy)
- Peter Liston (ACT Government)
- David Rush (ACT Parks & Conservation Service)
- Lloyd Van der Wallen (NSW National Parks & Wildlife Service)
- Sandie Jones (Univ. Of Canberra)
- Rod Pietsch (NSW National Parks & Wildlife Service)

Goal

- To guarantee that *Delma impar* can survive, flourish and retain its potential for evolutionary development in the wild.
- To prevent loss of genetic variation throughout the geographic range of the species.

Objective

- Develop a conservation strategy for conservation of *D. impar*, based on conservation of groups of populations throughout its geographic range.

Justification of Objective

Primary principles of biology conservation, both endorsed here for *D. impar*, are:

- i. specific habitat adaptations throughout the geographic range, as reflected in genetic differentiation between areas, should be preserved to ensure the evolutionary viability of the species;
- ii. Where a species has been fragmented into disconnected populations across its geographic range, populations should be protected across that range to guard against unpredicted events that eliminate populations in a region.

We need to ensure we maintain / do not lose genetic variation across the range of species. This may be achieved by identifying relative homogenous groups of populations and ensuring that these entities are maintained throughout time. The approach adopted in this document is to identify groups of populations, each here termed a "metapopulation" or "cluster", which are deemed to be adapted to the habitat characteristics of a region, and will contain the unique genetic material characteristic of that region. (Previous studies of *D. impar* suggest that there is little genetic variation between populations within a geographic region.) A cluster may contain a single population, but more commonly will consist of two or more populations. Each cluster needs to be conserved, primarily as an evolutionary important unit, but also as a guard against large-scale catastrophe. This approach satisfies both conservation principles above.

These tenets have been incorporated in an operational objective to ensure conservation of *D. impar*: "to ensure survival of all clusters with less than 1% risk of extinction of each metapopulation in 100 years." The premise is that the maintenance of a viable metapopulation for each defined geographic region will conserve the species and its incumbent genetic variation within that region, and metapopulations together will conserve the species.

Steps To Achieve Objective

- i. Delineate clusters of *D. impar* populations across its geographic range.
- ii. Describe the populations comprising each cluster; and their biological and management status.
- iii. Investigate the viability of each individual cluster, and of populations within clusters, using the Population Viability Assessment modelling approach.
- iv. Generate a series of explicit recommendations detailing the actions needed to conserve *D. impar*.

Section 1 Clusters

Criteria for formation of clusters

Clusters have been tentatively defined using geographic and environmental criteria to represent areas of similar habitat. Of the 17 potential clusters, we have information about the genetic composition of only three. There is a need to further investigate genetic variation between populations to ensure complete representation of the range of differentiation within the species. To complement this research, multi variate data analysis of habitat variables should be conducted to identify similar populations and to review the composition of clusters.

Potential Clusters

Victoria

1. North Melbourne, Craigieburn Volcanic Plains
2. West Melbourne, Keilor, Werribee Plains
- 3.a) Western Victorian West Volcanic Plains
- b) Western Victorian East Volcanic Plains
4. North Ballarat cluster, isolated volcanic areas in central Victoria
5. East Grampians (potential link to Western Volcanic Plains cluster)
6. Horsham cluster, Wimmera grey cracking clays
7. Northern alluvial plains
8. North-eastern slopes, Benalla (not volcanic)
9. North-eastern slopes, Alexandra (not volcanic)
10. Gippsland Plains (Bioclim, geographically isolated)
11. South east Bendigo, Mitiamo volcanic belt.

South Australia

12. Bool Lagoon, SA, grey cracking clays (may be extinct ?).

NSW

13. Cooma-Monaro Plains treeless basalt plains, *Stipa-Poa* community under stones
14. Gilmore (geographically isolated), Batlow (close to Gilmore, old record)
15. Yass
16. Goulburn (most northern population), Windellama Road (34 47S 149 44E, found in *Phalaris*, although native habitat is close by).

ACT

17. Gungahlin, Majura Valley, Jerrabomberra Valley and Yarramundi Reach, and new site at Sutton (close to ACT, but in NSW).

See also Figure 1 and Tables 4.1 and 4.2 at the end of this section

Section 2 Populations

For the purposes of this report, a population is defined as a group of individuals capable of interbreeding naturally.

The modelling approach used to assess the viability of the core conservation entities, the clusters, requires detailed information on all *D. impar* populations. This detail includes population size, area of habitat, land tenure and current management. The spatial location of populations is shown on Figure 1, and the information on each population can be found in Table 4.1 and 4.2. In many instances this information is incomplete and further research or monitoring is required to fill these gaps. Particularly important are improved detection techniques for the species, allowing more accurate population estimates of the species. This and other explicit recommendations can be found in Section 5.

In the absence of complete information for each population, certain population characteristics were estimated to permit the PHVA process to continue. Where this has been done the logic followed and the underlying assumptions have been included to provide a full explanation of the process. Two population characteristics of crucial importance to the modelling process were not known, i.e. population sizes and carrying capacity of individual areas of habitat. The rationale used to estimate these parameters is detailed below.

Population Estimates

Estimates of population size in entire patches were based on the numbers of individuals caught in traps within a patch. Consequently, it was necessary to determine the area surrounding a standard trapline from which the trapped animals came, i.e. the trap catchment area. With this information trap numbers could be converted to a density for the trap catchment area, and thence to a patch density by combining information from all traps in the patch. Two approaches were used to determine trap catchment areas:

- extrapolation from trap catch records, and
- extrapolation from animal movement characteristics.

a. Trapline catch records

An initial assumption was that a cross-pattern trapline (as used in the ACT) had a minimum catchment area of 0.25 ha. Data from the repeat trapping of the 20 ha AGSO site were used to estimate the maximum size. The first trapping of the site with two traplines yielded 5 animals per trap. Trapping effort was subsequently increased to 11 traplines. If the catchment area for a trapline was significantly larger than 0.25 ha, perhaps up to 10 ha, then the number of animals yielded per trapline by the second trapping effort would have been much less than 5, as animals from the new trapline areas would already have been removed by the first two traplines. Numbers of animals caught per trap in the second trapping were similar to those in the first, suggesting that the trapline catchment area was not much bigger than the extent of the trap, 0.25 ha. From this information trap catchment area was conservatively estimated to be 0.5 ha, allowing a degree of conservatism.

b. Animal movement characteristics

Again, the initial assumption was that a trapline had a minimum catchment area of 0.25 ha. Then information on the extent that *D. impar* move was used to estimate from how far outside the trapline area animals may move to encounter a trap. Records show that *D. impar* are relatively active small reptiles, with 20 m the approximate mean distance which individuals have been recorded moving. Consequently, because only half of the potential animals may move towards the trapline, an extra 10m was added around the trapline area, giving a trapline catchment area of 0.5 ha.

Patch densities from trap catchment densities

The concordance of these two approaches in their estimates of the trapline catchment area gives confidence that it is a reasonable approximation of the true catchment area. It should also be noted that both these approaches are based on the assumption that the trapline catches 100% of animals within its catchment area. This is unlikely to be correct, making all density estimates conservative.

Density estimates were obtained for each trapline within a patch, using the maximum number of individuals caught at each trap in any trapping period. Records for traps which caught no animals were also included. A mean was taken of these density estimates and applied to the total area of the patch; the traplines' locations being assumed to be representative of conditions in the patch. In many instances traplines were located randomly through the patch and so the representative assumption is valid. In other cases where there has been trapline positioning in the best areas, this will not be valid and populations will be overestimated.

Carrying capacity estimates

Carrying capacity (K) has not been measured for any *D. impar* population. This demographic characteristic is necessary for the PHVA process and so it had to be estimated. Two pieces of information provide some insight into K for the group of populations within a cluster; the maximum population density and the variability of population densities.

Minimum K

One of the assumptions underlying the grouping of populations within a cluster is that there is a degree of similarity in environmental and habitat conditions within that cluster. Consequently, although some populations

may have low densities, perhaps as a result of more intense predation or disturbance, all populations have the potential of matching the K in the cluster, given appropriate management. So the highest population density in a cluster has been used to calculate a K value for each population. This was deemed to be the minimum K value because density estimates were largely based on adult captures, with the survey method not sampling the whole population.

Maximum K

The variability of population densities in a cluster, and the relationship of this variability to habitat condition, can tell us how resource-limited populations are. If there is a strong correlation between density and habitat quality, this suggests that populations in the best quality habitat may be resource-limited and close to K. If conversely there is no such relationship, the populations may be predation or disturbance-limited, and not close to K. Inspection of this information for the ACT cluster revealed a weak relationship between population density and habitat quality, and on this basis the maximum value for K was taken at twice minimum K.

Techniques for managing small populations

We have introduced this topic because of the very nature of *D. impar* populations, which are small, fragmented and often close to urban situations. They are, therefore, more likely to be suffering from loss of populations through urban expansion or from pressures which act on small populations. It is likely that management for long term viability will involve some form of interventionist management. This may include translocations, supplementation (movement of individuals between populations), habitat restoration or head-starting to reduce juvenile mortality.

Our recommendation to carry out a trial translocation is based on the benefits it may bring (providing information on the potential success or otherwise of translocations) and the likely possibility that populations will be lost and animals may need to be salvaged from sites prior to habitat loss. Habitat loss may be caused by urban development, fire, flood or other significant catastrophe. This situation has already occurred at a number of sites. We do not advocate the haphazard movement of animals without initial investigation.

Two justifications of trialling this technique are:

- Too many translocations occur under pressure, as last ditch efforts. Refinement of these techniques before crisis situations arise may increase the chance of success.
- Considerable knowledge can be gained by attempting translocations, particularly on habitat requirements. The quality of knowledge is improved if the translocation is set up as an experiment.

Section iii *Assessment of the viability of individual clusters and populations*

Further modelling, via PHVA analyses, needs to be completed to assess the viability of clusters and refine management within them. However, we consider that the population and life-history data presently available are inadequate to enable any such analyses to be reliable indicators, and urge that these data are collected as a matter of priority.

Section iv *Recommendations for conservation of D. impar.* (Note: not in any priority order)

Research

1. Secure funding for the research outlined below.
2. Investigate and improve detection techniques (including capture, marking and tracking). This is critical for accurate data analysis and population assessment.
3. Investigate and document the statistical significance of trapping effort, i.e. be able to place confidence on limits of detection for a specified trapping effort.

4. Determine any genetic differences in populations (perhaps including DNA techniques such as mDNA, microsatellites, etc.) on a regional basis to support cluster decisions (at present there is genetic information for only three of the 17 clusters).
5. Conduct surveys to determine population size and extent of habitat of *Delma impar* at known sites, historic sites and new sites identified by Bioclim.
6. Conduct life-history data
7. Conduct multivariate data analysis of habitat variables to determine similarities of sites within and between clusters and aid in refining cluster boundaries.
8. Determine differences in site variables where the species is present and absent (including historical information).
9. Investigate source/sink populations and the capacity of movement and dispersal of individuals between sites within clusters.
10. Determine the species capacity for recolonization and the maximum distance over which this can be achieved unaided (time frame over which we can expect a population to recover from a disaster, genetic versus population).
11. Determine any behavioural aspects of the species which may affect carrying capacity (i.e. territoriality).
12. Investigate the carrying capacity of different habitat sizes and qualities, and the biological processes which may impact on carrying capacity.
13. Model viability of clusters via PHVA analysis

Management

14. Develop a GIS database of all sites/populations, cluster boundaries, habitat and genetic information. (Custodian - National Recovery Team).
15. Determine other conservation significance of sites which contain *D. impar* (cultural, flora and fauna).
16. Ensure the distribution of new research information to and from all stakeholders. This should be coordinated by the National Recovery Team.

Techniques for managing viable populations

17. Determine the viability of management options (such as translocation, supplementation, corridor use, habitat rehabilitation, head-start techniques) which may be used to manage and conserve a small population.
18. Develop criteria for translocation and supplementation projects which outline the situations where such management is needed, protocol for conducting these activities and test proposals with PHVA modelling (Recovery Team responsibility to facilitate the development of criteria for translocations).
19. Carry out a trial translocation to determine the likely success and preferred methodology.
20. Develop a systematic process of monitoring and ensure that enough animals are used to maximise chances of a successful translocation.
21. Monitor survival and reproductive output of translocated individuals.

Monitoring

22. Monitor key sites to determine effectiveness of conservation management strategies.
23. Monitor population size and structure to determine if any supplementation or movement of animals is required between populations.
24. Model (PHVA) and monitor impacts on populations from which animals are removed for supplementation to determine any adverse impacts on source populations.

Table 4.1 Cluster locations of *Delma impar*

Cluster Name	Soil Type	Significance	Number of pops	Delma pop	Area	Possible Populations	Av Dens	Topography
1. North Melbourne								
2. West Melbourne								
3a. Western Victorian West Volcanic plains		Isolated volcanic areas	7	860	1105	8	0.86	
3b. West Victorian East Volcanic plains			19	2516	891	21	5.67	
4. North Ballarat cluster			3		192	9		
5. East Grampians			7		309	8		
6. Horsham cluster,		Link to western plains	0		7	2		
7. Northern alluvial plains	Wimmera grey cracking clays		0		11	2		
8. North eastern slopes, Benalla			2		<3	3		
9. North eastern slopes, Alexandra		Not volcanic	5		300	6		
10. Gippsland Plains		Not volcanic	17		?	1		
11. South east Bendigo, Mittiamo volcanic belt South Australia		Geographically isolated	1		?	2		
12. Bool Lagoon New South Wales	Grey Cracking clays	Only SA population	0		60	0		
13. Cooma-Monaro			1		?	4		
14. Gilmore, Batlow	Basalt soils, rocky	only Monaro site			?	1		
15. Yass		Geographically isolated	1		90	2		
16. Goulburn		Geographically isolated	0		?	2		
		Most northern distribution	0		?	1		
17. ACT	Australian Capital Territory	Highest population	1		?	1		
			11	13591	1169	12	9.72	

Figure 4.1. Map showing distribution of *Delma impar* Striped Legless Lizard, throughout south-eastern Australia

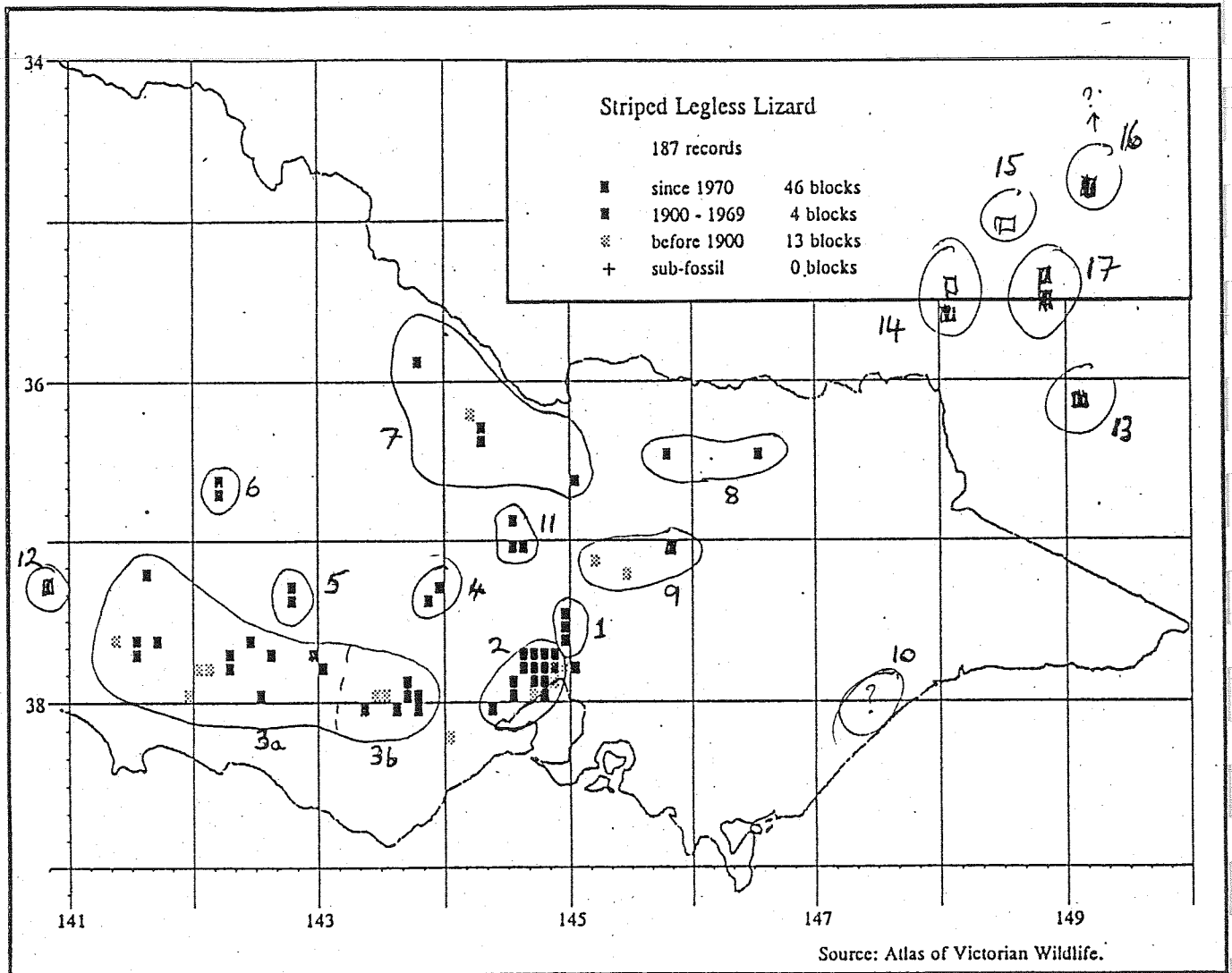
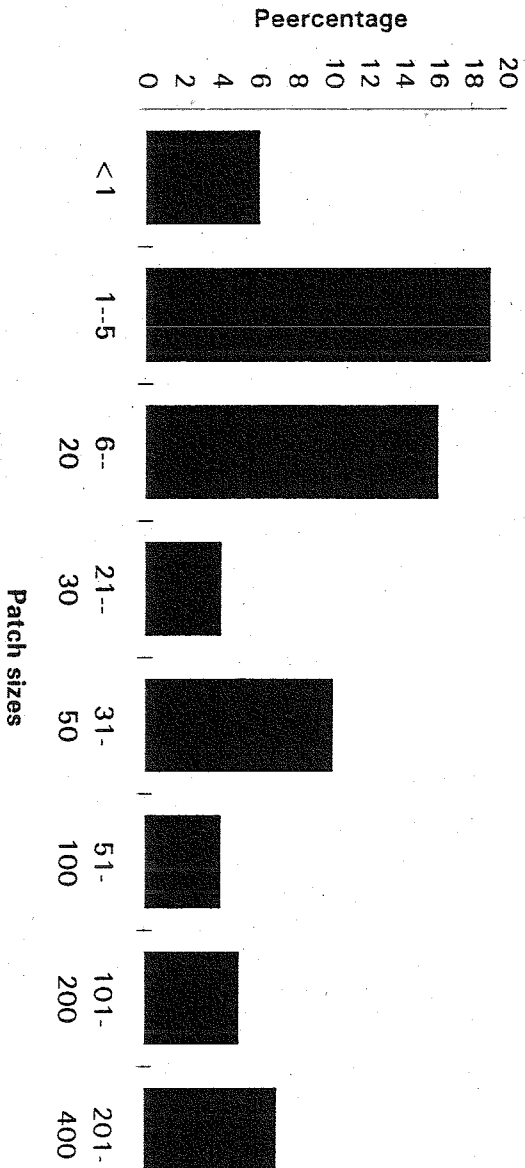


Table 4.2 Details of individual *Delma impar* populations (continued) plus graphical representation of figures (below)

Location	Year	Population	Area (ha)	Search Method	Number of Sites	Number of Individuals	Year	Status
Lake Elston	1996	Public Land	7624	1/150 rocks hand search	11	5	1990	N
Lake Epoufak	1996	Public Land	38 211 149 92	Hand search	16	1	1990	N
Slope Jug Lane	1996	Private Land	34 47 149 44	Majora 1-5, Mitchell old 1, 2	16	173	2000	N
Rieseddie	1996	Private Land	8727 002938	GTC 4, 5, FFA 1-5, MUI 1	26	190	1950	N
Digpen Monument, Mia Mia	1996	Private Land	8727 938992	Wells Stn, 1-6, Kerry 1-4, Mitchell New	10	42	1750	N
Bool Lagoon	1996	Private Land	8727 916012	Mitchell old, Pal 1-6, CSIRO 1-2, ZCV 1-4, Hoskins, Cam	13	291	4225	N
Coona	1996	Private Land	8727 938998	Kaleen 1-6	9	31	63	N
Abable	1996	Private Land	8727 933001	PAAF, TP 1-2, Stuart 1-2, Grace 1-3, Stockpound	14	71	2940	N
Ginnole	1996	Private Land	8727 965850	Kosc. 1-2, Pal 1 (pre 1994 old)	5	14	100	N
Ballow	1996	Private Land		Bonshaw 1, 3, Harman 1-5, Woden 1, 5, 6, Douglas	2	14	400	N
Yass	1996	Private Land		AGSO 1-11 (developed) - no longer exists	4	16	69	N
Goulburn	1996	Private Land		Site 2, 3	3	16	45	N
Majora Valley	1996	Private Land			5	50	50	N
Marangari	1996	Private Land			5	50	50	N
Kerny	1996	Private Land			5	50	50	N
Crace/Gungahleria	1996	Private Land			5	50	50	N
Kaleen	1996	Private Land			5	50	50	N
South Mitchell	1996	Private Land			5	50	50	N
Koocumbra	1996	Private Land			5	50	50	N
Jeakoberra Valley	1996	Private Land			5	50	50	N
AGSO	1996	Private Land			5	50	50	N
Mussum	1996	Private Land			5	50	50	N
Sutton NSW	1996	Private Land			5	50	50	N
Gundaroo (misc)	1996	Private Land			5	50	50	N
Melino	1996	Private Land			5	50	50	N
Gundaroo	1996	Private Land			5	50	50	N
Stewarston	1996	Private Land			5	50	50	N

Distribution of habitat sizes



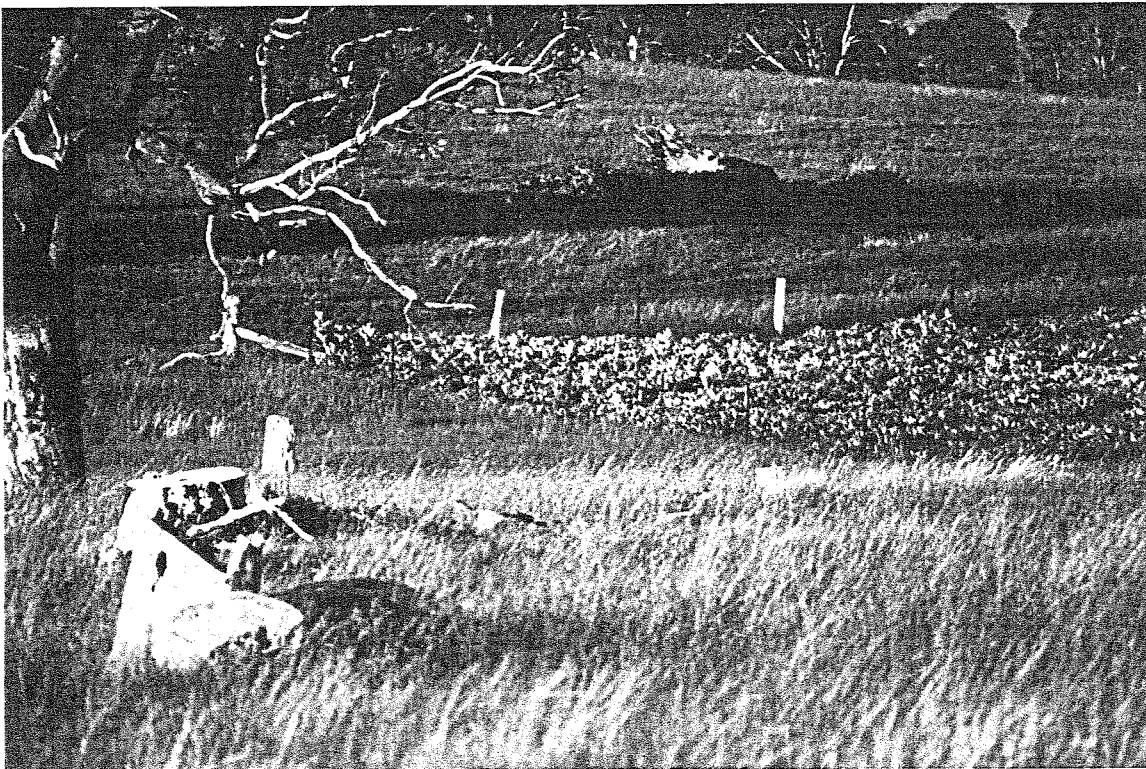
DELMA IMPAR
Striped Legless Lizard

Population & Habitat Viability Assessment Workshop

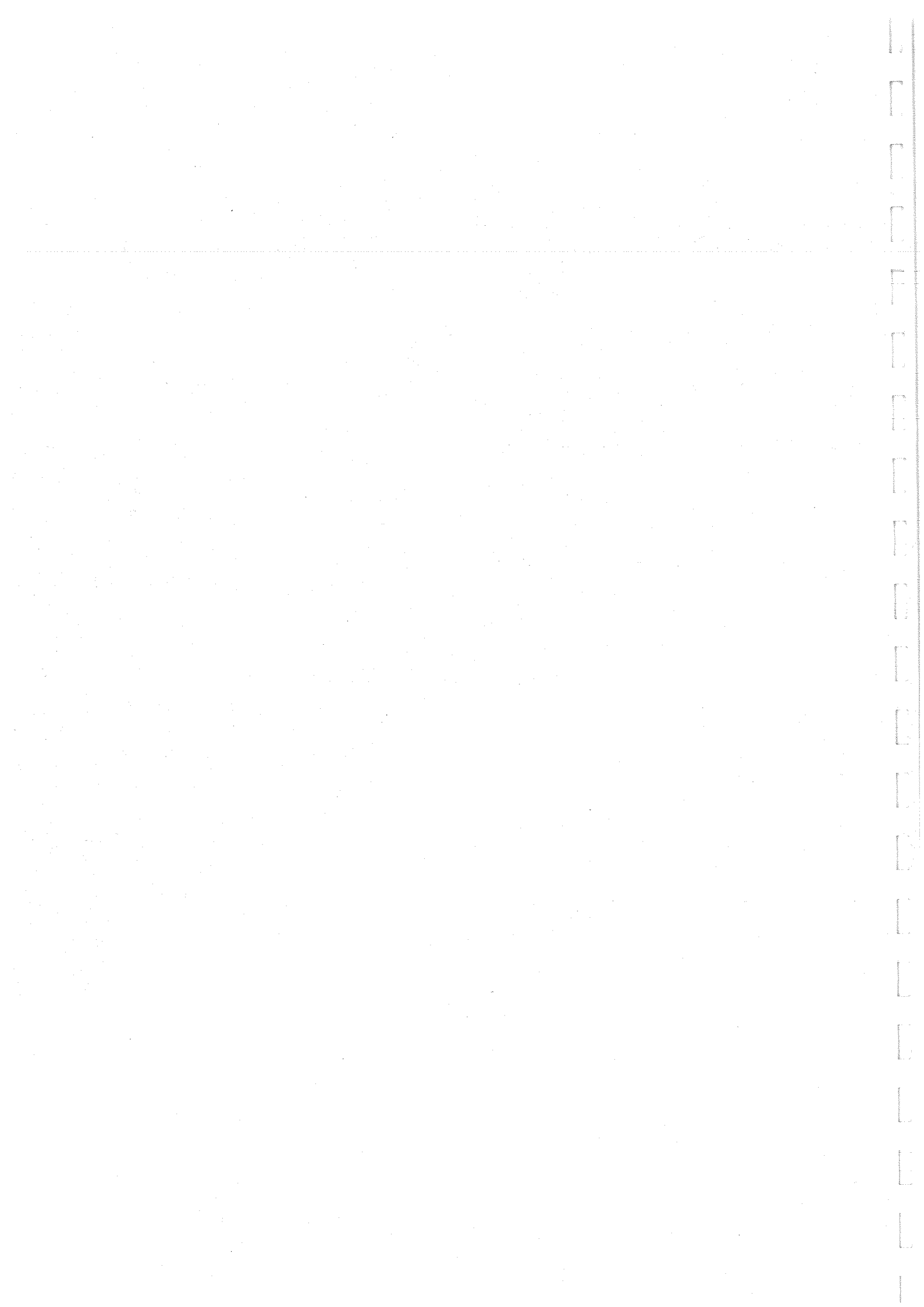
Canberra

30 July - 2 August 1996

FINAL REPORT



Section 5 ~ **Modeling**



Population biology and modelling of the Striped Legless Lizard *Delma impar*

Working Group Participants

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Introduction

The need for, and consequences of intensive management strategies can be modelled to suggest which practices may be the most effective in conserving the Striped Legless Lizard. VORTEX, a simulation software package written for population viability analysis, was used as a tool to study the interaction of a number of life history and population parameters treated stochastically, to explore which demographic parameters may be the most sensitive to alternative management practices, and to test the effects of a suite of possible management scenarios.

The VORTEX package is a Monte Carlo simulation of the effects of deterministic forces, as well as demographic, environmental and genetic stochastic events on wild populations. VORTEX models population dynamics as discrete sequential events (e.g. births, deaths, sex ratios among offspring, catastrophes, etc.) that occur according to defined probabilities. The probabilities of events are modelled as constants or as random variables that follow specified distributions. The package simulates a population by stepping through the series of events that describe the typical life cycles of sexually reproducing, diploid organisms.

VORTEX is not intended to give absolute answers, since it is projecting stochastically the interactions of the many parameters which enter into the model and because of the random processes involved in nature. Interpretation of the output depends upon our knowledge of the biology of the Striped Legless Lizard, the conditions affecting the population and possible changes in the future.

The working group approached the modelling of populations of *D. impar* in the following ways:

Identifying critical life history parameters

What little data exists on the population biology of Striped Legless Lizard was compiled to establish an initial, baseline population model. The sensitivity of that model to varying some of the more uncertain parameters was tested, to identify those parameters for which more accurate data is most critically needed.

Assessing threats to remaining populations

Beyond further loss of habitat, a number of threats were deemed to be of potential importance to the remaining populations, and were evaluated for likely impact on various of the life history parameters of *D. impar* (e.g. increased juvenile mortality) by the Habitat & Threats working group (Section III). These included fire, drought, grazing and predation. The relative importance of these threats was assessed by comparing the impact of varying the relevant life history parameters in the VORTEX simulations. Specific models were established to approximate the characteristics of *D. impar* populations in the ACT area under conditions approximating all possible combinations of drought, grazing, and fires of varying intensity.

In addition to the threat resulting from loss of habitat (i.e. reduced population size), the incidence of fire and rate of predation emerged as likely critical threat factors. The impact of these threats was further examined.

The impact of fire management strategies

The potential for 'control burn' fire management strategies to enhance viability of *D. impar* populations was assessed using a model based on the characteristics of the generalised baseline population. The impact of control burns of varying frequency was compared with the potential effect of unmanaged wildfire.

The impact of predation on urban populations

The relative impact of predation on *D. impar* populations seemed likely to be somewhat site-specific, varying depending upon the presence and abundance of various potential predator species. Specifically, urbanisation may increase the abundance of both feral predators, such as cats, and native avian predators. Populations of *D. impar* abutting urban areas (as with many ACT populations) may therefore face significant additional threat. To test this, extensive survey data were re-assessed to determine possible differences in predation rates between sites characterised as 'rural' and 'urban'. This involved a comparison of tail autotomy rates, and age structures between rural and urban sites.

5.1 Baseline Population Model for *D. impar*

5.1.1 Input Parameters for Simulations

Mating System: polygynous. This is assumed on the basis of known mating systems of other *pygopodids*.

Age of First Reproduction: for this species VORTEX precisely defines breeding as the time at which eggs are laid, not simply the age of sexual maturity. In addition, the program uses the mean (or median) age rather than the earliest recorded age of young production.

- Females were assumed to be reproductively capable at 3 yrs. There is virtually no data on age-specific fertility in *D. impar*, however it is known that it is possible for four year old females to breed. A single animal was initially caught and at approximately one year of age (an estimate based on a comparison of snout-vent length, SVL, with records of SVL of ten known-age animals incubated in captivity in Victoria and ACT). The female was re-caught in gravid condition in the ACT three years later (i.e. aged ~4yrs), and subsequently laid eggs in captivity (Banks pers. comm.).
- Males were assumed to be reproductively capable at 2 yrs. This was based entirely on evidence from other lizard species, which indicating that males tend to reach sexual maturity one year earlier than females.

Age of Reproductive Senescence: reproductive longevity is defined as the age at which breeding ceases and VORTEX assumes that animals can breed (at the normal rate) throughout their adult life. Initial estimates of 10 yrs were used in a baseline model. However, total longevity of up to 20 yrs has been recorded in gekkonids. Therefore, reproductive senescence of 10 yrs could be an under-estimate for *D. impar*.

A range of values was subsequently tested (10, 15, 20 yrs) for impact on the overall model.

Sex Ratio at Birth: in absence of evidence of skewed sex ratios, it was assumed that the sex ratio at birth was even. Samples from only two groups have been definitively sexed, using x-ray sexing techniques, and no consistent trend of skewed sex ratio was suggested; (the Ravenhall Prison site produced 3:10 (males: females) and the Sunshine Tip site produced 4:2 - Banks unpublished data).

Offspring Production: for the purposes of modelling all population dynamics, we defined "reproduction" for a given female as the successful laying of eggs.

It was assumed that all gravid females had SVL > 72 mm (ACT P&CS survey '89 - '95). Therefore, half of the animals above that size were presumed to be females, of those 136/331; ie: ~50% were gravid. This is based on the following assumptions:

- equal sex ratio for captures
- all suspected gravid females are actually gravid (distended body size).

Therefore, upper and lower limits of 50% - 100% for female reproductive success can be determined. Reproductive success was set at the upper limit (100%) of this expectation in the baseline models, on the basis of comparisons with gekkonid spp. Subsequent sensitivity analyses tested the effect of a range of reproductive success values.

Annual variation in female reproduction is modelled in VORTEX by entering a standard deviation (SD) for the proportion of females that do not reproduce in a given year. Since no appropriate data were available for this species, we set this variation to approximately 25% of the mean value. VORTEX then determines the proportion of females breeding each year of the simulation by sampling from a binomial distribution with the specified mean (e.g. 20%) and standard deviation (e.g. 5%).

Based on data from eggs laid by captive animals, and data from other pygopodids, two eggs/clutch were assumed for all clutches.

Male Breeding Pool: no data are available for this parameter, and all males of reproductive size/age (ie. older than two years) have been assumed to be available for breeding in each year.

Mortality: again, data are lacking on the mortality of specific age-sex classes. However, we have assumed the following mortality rates:

0-1 yrs - 50%
 1-2 yrs - 20%
 2 - adult - 10%

As with the environmental variation set for female reproduction, we set the annual variation in mortality to be approximately 25% of the mean rates.

Catastrophes: catastrophes are singular environmental events that are outside the bounds of normal environmental variation affecting reproduction and/or survival. Natural catastrophes can be tornadoes, floods, droughts, disease or similar events. These events are modelled in VORTEX by assigning a probability of occurrence and a severity factor ranging from 0.0 (maximum or absolute effect) to 1.0 (no effect).

In the baseline model, the effects of two catastrophe types were modelled:

- Fire:

Frequency:	4/100yrs
Impact on reproduction:	0.2
Impact on survival:	0.5

- Drought:

Frequency:	10/100yrs
Impact on reproduction:	0.2
Impact on survival:	0.9

Later models attempted a more sophisticated analysis of varieties of catastrophes.

Initial Population Size: the Gungahlin population was used as a basis for the initial baseline scenario. This was estimated as 2,000 individuals, assuming population density of 30/ha (published estimates of between 10 - 40/ha) over approximately 70ha.

Carrying Capacity: the carrying capacity, K, for a given habitat patch, defines an upper limit for the population size. Above this upper limit, additional mortality is imposed across all age classes in order to return the population to the value set for K. VORTEX, therefore, uses K to impose density-dependence on survival rates. The program also has the capability of imposing density-dependent effects on reproduction that change as a function of K, but since no such data are available for Striped Legless Lizard populations, we chose not to include density-dependent reproduction in our models.

In the absence of hard data, initial models incorporated a carrying capacity of double the current population size.

Iterations and Years of Projection: all scenarios were simulated 100 times, with population projections extending for 100 years. Output results were summarized at 10-year intervals for use in some of the figures that follow. All simulations were conducted using VORTEX version 7.2 (June 1996).

5.1.2 Results

Simulations using the baseline model data indicated a population with a stochastic growth rate of 6.2%, zero probability of extinction over the next 100 years given maintenance of current conditions, and resulting in a mean population size at the end of 100 years of approximately 3,200 individuals (Table 5.2; filename DELMA101.OUT).

There is, however, a high degree of uncertainty regarding some of values used as baseline parameters, and comparison between the model output and capture data indicates the extent to which this may impact on resulting population projections.

The baseline model parameters result in a mean final population age structure of 1 yearling to every 3 adults (non-yearlings). However, capture data suggests a quite different age structure to the population. Age classes in wild populations have been inferred from data on body size (mean snout-vent length, SVL) from animals trapped in ACT and Victoria. The limited data available from captive and wild populations suggest an age/body size relationship as follows:

Hatching - 1 yr:	< 60mm SVL (2 ACT captive hatchlings; 3 Victorian captive hatchlings)
1 - 3 yrs:	60 - 75mm SVL
over 3 yrs	> 75mm SVL (assumed gravid females)

Interpreting trapping data from ACT and VIC in the context of these age/size classes suggests a substantially greater representation of adults *D. impar* populations:

- Vic data 1:11.5 yearlings: non-ylings (n = 411) (Banks unpublished Victorian survey data)
- ACT data 1:14.5 yearlings: non-ylings (n = 929) (ACT P&C unpublished survey data '89 - '95)

This is substantially different from the age structure resulting from simulations using the model parameters, and suggests that the baseline data may be deficient in any of the following ways:

- over-estimated female reproductive success rate
- under-estimated yearling mortality rates
- over-estimated adult mortality rates
- under-estimated longevity

Conversely, the sampling techniques may not adequately measure age-structure.

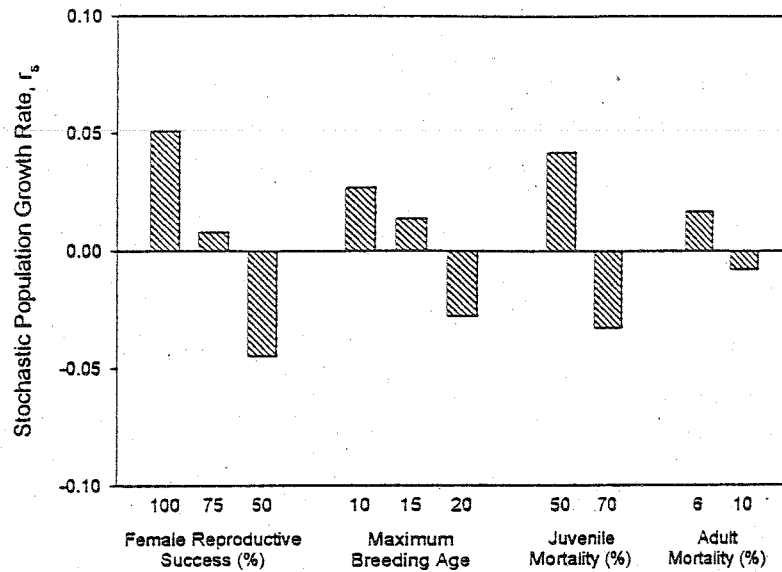
Sensitivity Analyses

The efficacy of the sampling techniques could not be tested in the context of the workshop, however the effect of modifying reproductive success, longevity and mortality schedules was tested using the following range of values:

Reproductive success:	100%, 75%, 50%
Longevity:	10yrs, 15yrs, 20yrs
Age-specific mortality - mortality rates for 0 - 1 yrs and for adults were varied.	
0 - 1 yrs	50%, 70%
adult	10%, 6%

Each value was tested against all possible combinations of the other variables. All other parameters remained as baseline values.

Detailed results for the simulated effect of varying each of the critical baseline parameters are given in Tables 5.2 - 5.5. The average effect of each value tested is indicated through mean stochastic growth rates calculated across all simulations including that variable (Fig. 5.1 & Table 5.1).

Figure 5.1 *D. impar* population viability: sensitivity to variation in life history parameters

In addition, the relative effect of change in each parameter is indicated through interpolating the degree of change in mean stochastic growth rate (r) resulting from change in unit measure (ie: 1% or 1yr) (Table 5.1). Variations in longevity and adult mortality rates result in the greatest unit change, with a change in growth rate of $\sim 0.6\%$ for each year longevity estimates are extended, and for each 1% change in adult mortality rate.

Table 5.1 Simulated effect on population growth rate of varying critical *D. impar* life history parameters

PARAMETERS	STOCHASTIC GROWTH RATE	Δr /UNIT CHANGE
Reproductive success (female)		
100%	0.051	
75%	0.008	
50%	-0.045	0.0019
Longevity		
10yrs	-0.028	
15yrs	0.014	
20yrs	0.027	0.0055
Juvenile mortality		
50%	0.042	
70%	-0.027	0.0038
Adult Mortality		
10%	-0.008	
6%	0.017	0.0062

The degree of variation in population predictions that may result from uncertainty in these parameters can be indicated through considering 'best case' and 'worst case' scenarios that result from sensitivity testing across all parameters.

'Worst case' scenario:

0-1 yr mortality rates:	70%
adult mortality rates:	6%
Longevity:	10 yrs
Reproductive success:	50%

'Best case' scenario:

0-1 yr mortality rates: 50%
 adult mortality rates: 6%
 Longevity: 20 yrs
 Reproductive success: 100%

Stochastic growth rates for the 'worst case' scenario are -13.2% and probability of extinction of a population of 2,000 animals is predicted to be 100%. In contrast, the 'best case' scenario predicts growth rates of 11.8%, with a predicted 0 probability of extinction.

Table 5.2: 50% 0-1yr mortality; 10% adult mortality

Longevity	File name	Repro. Success	Growth rate		Prob. Extinction	Pop. Size		Het	Time to Extinction	
			r	SD(r)		N	SD(N)		Median	Mean
10 yrs	DELMA101.OUT	100%	0.062	0.217	0.000	3248.73	1089.70	99.24	0	0.0
	DELMA102.OUT	75%	0.013	0.211	0.010	2391.78	1341.54	98.39	0	81.0
	DELMA103.OUT	50%	-0.058	0.225	0.430	105.51	177.33	83.88	0	76.3
15 yrs	DELMA104.OUT	100%	0.086	0.215	0.000	3456.49	922.06	99.43	0	0.0
	DELMA105.OUT	75%	0.048	0.198	0.000	3128.88	1051.19	99.36	0	0.0
	DELMA106.OUT	50%	-0.014	0.203	0.010	959.60	1072.31	94.89	0	85.0
20 yrs	DELMA107.OUT	100%	0.091	0.217	0.000	3551.26	761.29	99.45	0	0.0
	DELMA108.OUT	75%	0.048	0.211	0.000	3199.83	1083.22	99.34	0	0.0
	DELMA109.OUT	50%	-0.006	0.207	0.020	1328.57	1243.98	97.13	0	87.5

For all results tables, r = mean instantaneous growth rate; SD(r) = standard deviation of r; N = mean final population size; SD(N) = standard deviation of N; Het = mean heterozygosity retained

Table 5.3: 50% 0-1yr mortality; 6% adult mortality

Longevity	File name	Repro. Success	Growth rate		Prob. Extinction	Pop. Size		Het	Time to Extinction	
			r	SD(r)		N	SD(N)		Median	Mean
10 yrs	DELMA110.OUT	100%	0.083	0.217	0.000	3525.90	823.87	99.40	0	0.0
	DELMA111.OUT	75%	0.034	0.208	0.000	2870.50	1128.62	99.10	0	0.0
	DELMA112.OUT	50%	-0.032	0.207	0.130	424.53	692.88	91.27	0	74.9
15 yrs	DELMA113.OUT	100%	0.108	0.217	0.000	3639.91	694.44	99.48	0	0.0
	DELMA114.OUT	75%	0.064	0.210	0.000	3373.09	899.09	99.44	0	0.0
	DELMA115.OUT	50%	0.011	0.203	0.000	1998.99	1380.23	98.57	0	0.0
20 yrs	DELMA116.OUT	100%	0.118	0.211	0.000	3674.72	666.56	99.51	0	0.0
	DELMA117.OUT	75%	0.077	0.205	0.000	3406.46	904.94	99.50	0	0.0
	DELMA118.OUT	50%	0.029	0.191	0.000	2781.59	1278.27	99.16	0	0.0

Table 5.4: 70% 0-1yr mortality; 10% adult mortality

Longevity.	File name	Repro. Success			Prob. Extinction	Pop. Size		Het	Time to Extinction	
			r	SD(r)		N	SD (N)		Median	Mean
10 yrs	DELMA119.OUT	100%	-0.032	0.227	0.160	419.18	671.83	89.38	0	82.2
	DELMA120.OUT	75%	-0.090	0.247	0.870	14.54	9.65	70.57	74	70.0
	DELMA121.OUT	50%	-0.132	0.245	1.000	0.00	0.00	0.00	53	53.4
15 yrs	DELMA122.OUT	100%	0.008	0.209	0.000	1931.61	1371.98	97.83	0	0.0
	DELMA123.OUT	75%	-0.030	0.207	0.090	387.30	592.12	92.75	0	78.7
	DELMA124.OUT	50%	-0.085	0.232	0.780	15.86	15.14	69.00	79	71.3
20 yrs	DELMA125.OUT	100%	0.020	0.206	0.000	2565.66	1350.95	99.00	0	0.0
	DELMA126.OUT	75%	-0.015	0.200	0.030	1010.34	1183.29	96.25	0	82.3
	DELMA127.OUT	50%	-0.061	0.213	0.410	52.08	73.97	83.45	0	77.1

Table 5.5: 70% 0-1yr mortality; 6% adult mortality

Longevity.	File name	Repro. Success			Prob. Extinction	Pop. Size		Het	Time to Extinction	
			r	SD(r)		N	SD (N)		Median	Mean
10 yrs	DELMA128.OUT	100%	-0.005	0.210	0.000	1279.48	1275.64	96.72	0	0.0
	DELMA129.OUT	75%	-0.057	0.223	0.450	98.56	121.92	84.28	0	78.1
	DELMA130.OUT	50%	-0.118	0.240	0.990	4.00	0.00	56.25	56	58.0
15 yrs	DELMA131.OUT	100%	0.032	0.208	0.000	2800.54	1264.85	99.20	0	0.0
	DELMA132.OUT	75%	-0.007	0.201	0.010	1255.46	1281.61	97.39	0	81.0
	DELMA133.OUT	50%	-0.051	0.206	0.290	130.17	257.32	83.52	0	78.6
20 yrs	DELMA134.OUT	100%	0.043	0.212	0.000	3072.30	1066.26	99.35	0	0.0
	DELMA135.OUT	75%	0.013	0.198	0.000	2304.44	1271.65	99.01	0	0.0
	DELMA135.OUT	50%								

5.2 Assessment of Threats: ACT Populations

5.2.1 Model parameters

Specific models were developed to approximate the characteristics of *D. impar* populations in the ACT. Each population was built upon a series of general population types defined by the criteria outlined below.

Rural. Rural populations can be further subdivided by the following characteristics:

- *Grazed / Non-grazed.* Habitats subjected to grazing were assumed to have a larger environmental variance in *D. impar* reproduction due to random variations in food quality and availability in these areas. Whereas the baseline level of EV for the proportion of females breeding annually was set at 10%, this variation was increased to 25% in grazed areas. In addition, variation in the annual carrying capacity of grazed habitats was included by setting the SD in K to approximately 10% of the actual carrying capacity value appropriate for the particular population size modelled (see below).
- *Presence / Absence of Controlled Burn Management (Non-grazed areas only).* In those areas not subjected to grazing, grassland management can be effected by controlled burns, slashing, etc. It was assumed that this level of active management would essentially eliminate the occurrence of natural wildfires in the same area, although this assumption is open to further discussion. In rural areas, controlled burns were assumed to occur at three-year intervals, to reduce the proportion of females breeding annually by 10%, and reduce the survival of all individuals by 5%.

Urban. Urban populations can be further subdivided by the following characteristics:

- *Presence / Absence of Controlled Burn Management (Non-grazed areas only)*. In those areas not subjected to grazing, grassland management can be effected by controlled burns, slashing, etc. It was assumed that this level of active management would essentially eliminate the occurrence of natural wildfires in the same area, although this assumption is open to further discussion. Controlled burn management has the same characteristics of frequency and severity as those in rural areas.

Both rural and urban areas were modelled as either small or large populations. The Kosciusko site, with an initial population size of 100 and a carrying capacity of 520, was used for the small population example. For the large population type, the Kenny site was used. This site was assigned an initial size of 1750 animals and a carrying capacity of 4550 (for more information on the choice of initial population and carrying capacity, see the detailed discussion in the Population Working Group Report).

Additionally, each population was modelled with or without the impact of catastrophes. Each type of population had its own combination of catastrophic events, depending on site characteristics. The following were identified:

Drought

Grazed: 4% frequency; 100% reduction in reproduction, 5% reduction in survival

Ungrazed: 4% frequency; 95% reduction in reproduction, 2.5% reduction in survival

Wildfire

Rural: 4% frequency; 80% reduction in reproduction, 40% reduction in survival

Urban: 100% frequency; 10% reduction in both reproduction and survival

Severe fire in the presence of drought

Grazed: 1% frequency; 100% reduction in reproduction, 10% reduction in survival

Ungrazed: 2% frequency; 100% reduction in reproduction, 10% reduction in survival

Flood

All types: 4% frequency; 90% reduction in reproduction, 5% *increase* in survival

All combinations of population size and type produced a base set of 20 models. An additional 20 models were run that specifically investigated the impact of the choice of carrying capacity in each population size studied. This was done by assigning the carrying capacity as equivalent to the initial population size. Finally, specific models were developed to look at the sensitivities of these populations to juvenile mortality; these models will be described in more detail below.

5.2.2 Model Results

The results from each of the rural models for which the initial population size is below carrying capacity are shown in Table 5.6. Immediately apparent from these analyses is the fact that all population growth rates are positive; in other words, each of the simulated populations has the capability for population growth of between 3% and nearly 9% annually. Consequently, the probability of population extinction was essentially negligible for all populations modelled, and the final population sizes at the end of the 100-year simulation were all near the corresponding carrying capacity.

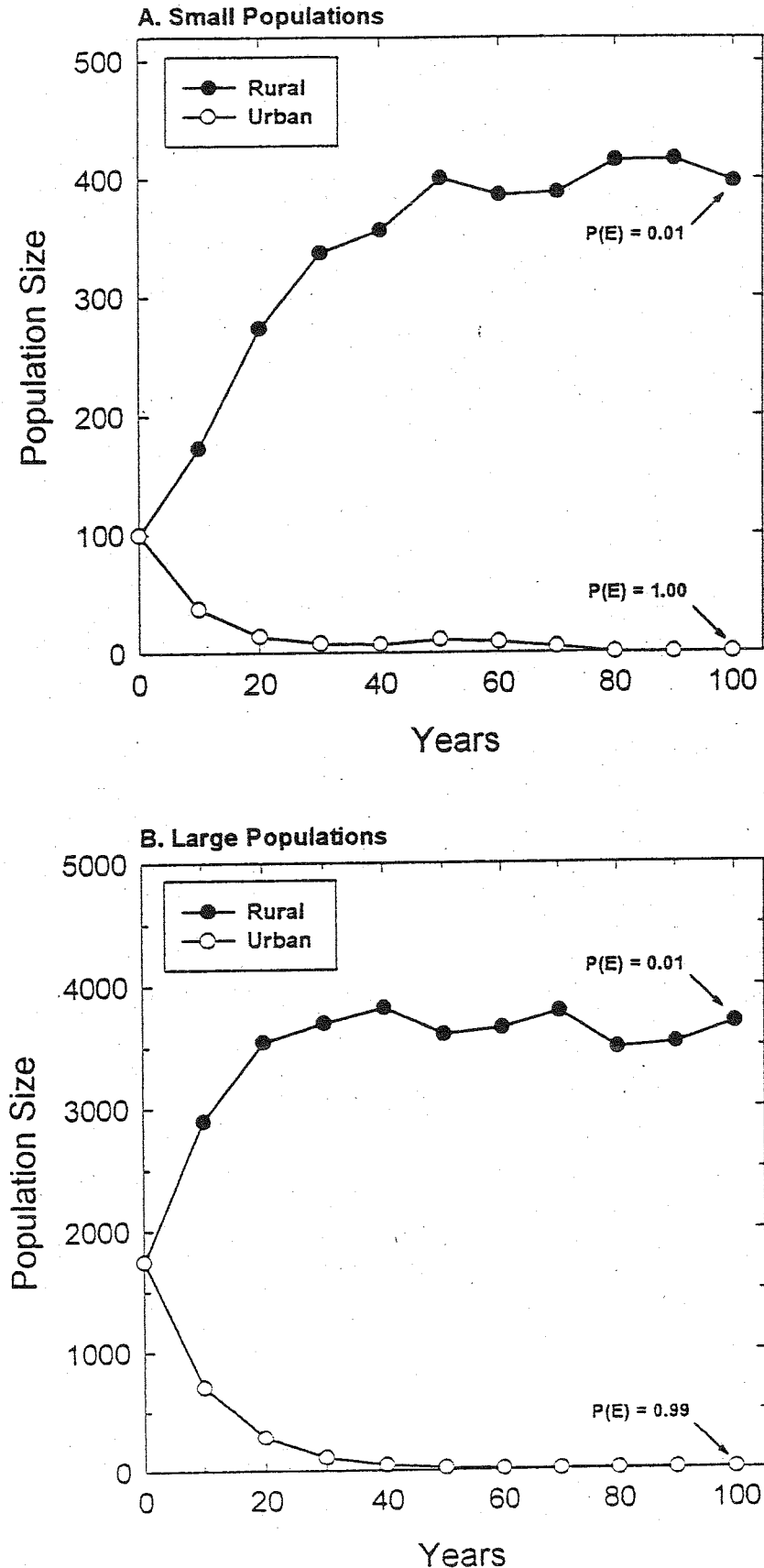
The impact of the modelled catastrophes is readily apparent by comparing a given scenario pair in which the catastrophes are included and then subsequently removed. In all cases shown in Table 5.6, the removal of catastrophes results in a stochastic population growth rate that is at least double that in the presence of catastrophes. Moreover, it appears from these analyses that the controlled burn management strategies in non-grazed habitats lead to lower growth potential of *D. impar* populations when compared to grazed habitats subjected to natural burns (for example, compare results in Files 401.OUT and 405.OUT).

Overall, under the conditions laid out in these models, rural populations of *D. impar* appear to be relatively secure from extinction. It is important to remember, however, that these conclusions are based solely on the conditions of the particular models described here and may be overly optimistic. If mortality or fecundity parameters are different in natural populations from those modelled here, it is possible that a very different picture could emerge. This will be explored in more detail later in this discussion.

Compared to their rural counterparts, urban populations show considerably greater instability and susceptibility to extinction (Fig. 5.2a & b). Only one of the eight urban population models shows a positive stochastic growth rate, and only large areas managed by controlled burns are relatively safe from extinction (Table 5.7). Once again, the amelioration of catastrophes is seen as having considerable benefit, as demonstrated by the significant increase in stochastic population growth rates. Additionally, the use of controlled burn management techniques as modelled here

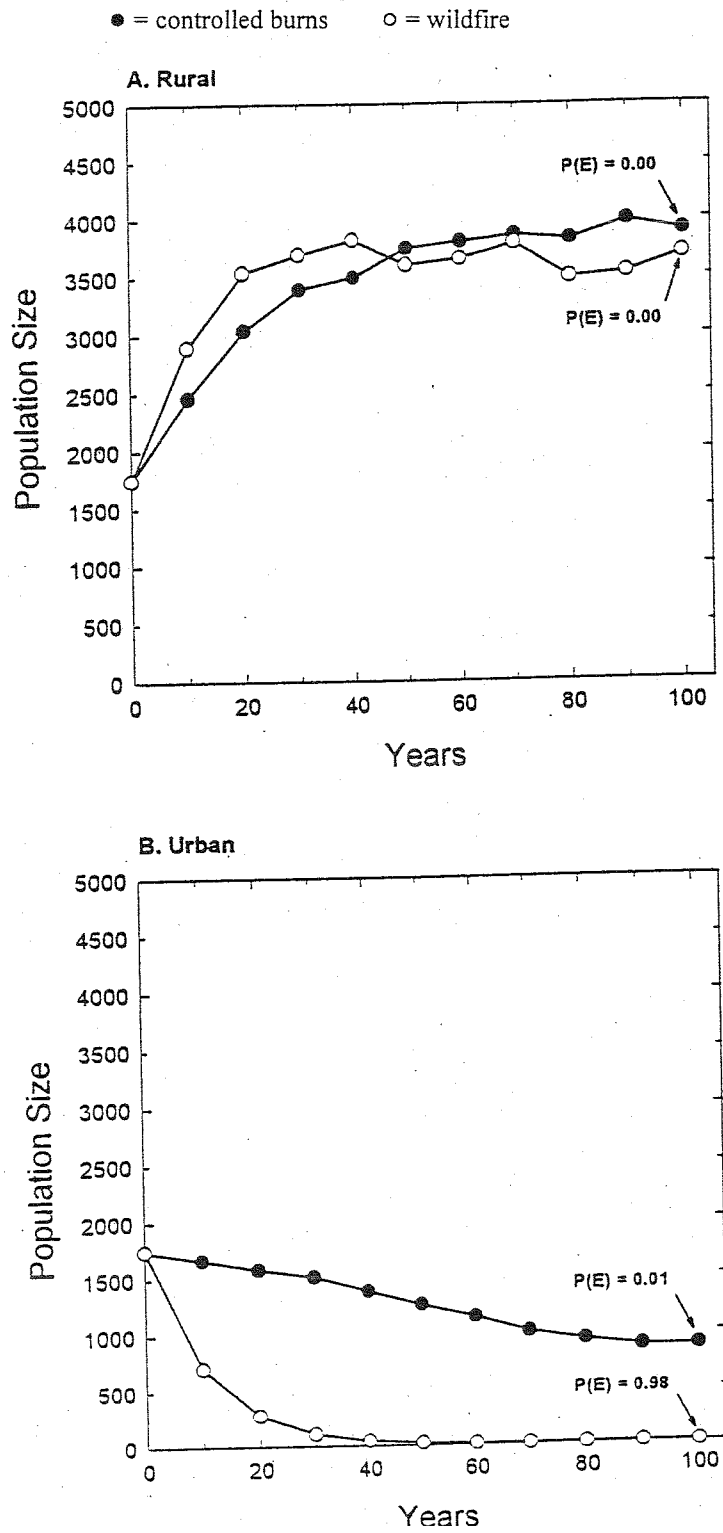
seems to have very positive relative effects on population dynamics. For example, large populations managed by controlled burn techniques (File 417.OUT) show a 1.3% rate of population decline, while those same populations subjected to annual wildfires without controlled burn management (File 419) decline at a rate of nearly 11% per year. As expected, the differences in the likelihood of extinction are dramatic between these two population types.

Figure 5.2a & b: Comparison of population size and viability in simulated populations of *D. impar* in 'rural' and 'urban' conditions



An interesting and important result from these analyses relates to the relative impact that controlled burn management techniques have on populations of *Delma impar* in rural and urban habitats. In rural areas, controlled burn management actually reduces the population growth potential compared to a non-management scenario where rural wildfires are allowed to occur, while the same techniques applied to urban areas result in higher *D. impar* growth rates compared to the situation where annual urban wildfires occur. The major difference between the outcomes of controlled burn management techniques between rural and urban population types appears to rest primarily in the nature of wildfires in these two areas: rural fires are quite severe but thought to be relatively infrequent, while urban fires may be relatively less severe but occur every year in some portion of a site. This serves to effectively reduce annual survivorship across the site. Thus, where the frequency of urban wildfires can be reduced by controlled burning regimes, the probability of extinction of *D. impar* populations may be dramatically reduced (Fig. 5.3).

Figure 5.3a & b: The simulated effect of wildfire and controlled burning regime on population size and viability in 'urban' & 'rural' *D. impar* populations



Response times to fires within the urban and peri-urban areas of the ACT are known to be short and rarely does a fire in such an area burn more than a few hectares. Therefore, the impact of such fires, although frequent, may be much less than the assumptions fed into the model. For example, a decrease of adult survivorship by 20% instead of 40% may be more realistic in these circumstances.

Simulation results are changed relatively little if these populations are in fact very close to their carrying capacity (Tables 5.8 and 5.9). Stochastic population growth rates remain positive for all rural populations and extinction risks remain low, although small populations that stay small throughout the 100 years of the simulations have a slightly higher risk of extinction due to stochastic demography. Urban populations remain highly destabilised, and in fact small populations under controlled burn management (Files 433 and 434) show consistently higher extinction risks than those allowed to grow beyond the initial size (Files 413 and 414).

As discussed above, the relative security of the rural populations displayed in this analysis is largely a function of the nature of the life table constructed for the models. The general sensitivity analysis described earlier in this section clearly demonstrated that relatively small changes in fecundity and/or mortality can have a major impact on the growth dynamics of simulated populations of *D. impar*. To assess the sensitivity of these particular populations to uncertainty in a single demographic parameter—juvenile mortality—all rural populations were repeated with this parameter increased from 50% to 60%. The results of these analyses are presented in Tables 5.10 and 5.11. It is clear that this relatively small change in mortality of a single age class has a considerable impact on the growth dynamics of these populations. For example, the small grazed population in the presence of catastrophes (File 501) shows a stochastic growth rate that is reduced by 90% over the corresponding scenario with 50% juvenile mortality (File 401). Even more significant is the observation that under higher juvenile mortality, small non-grazed areas subjected to periodic controlled burns (File 505) show a population decline at a rate of more than 1% annually, a dramatic effect compared to the 3% annual growth exhibited by the same population (File 405) with 50% juvenile mortality. As expected, these reductions in population growth directly lead to increased risks of extinction during the timeframe of the simulation.

In general, it appears that a lowered carrying capacity has the most pronounced effect on declining populations, resulting in slight additional reductions in growth but, more importantly, considerable increases in the risk of extinction. As an example of this observation, consider a small non-grazed area subjected to rural wildfires (File 507). If the population is allowed to grow to carrying capacity, the annual growth rate is -0.001 and the risk of extinction is 17%. However, if the population is already at carrying capacity, the growth rate becomes -0.013 and the extinction risk jumps to 47%. This increased risk results simply from the fact that the population remains small throughout the simulation and therefore is more susceptible to the stochastic forces that serve to destabilise it.

All in all, the brief sensitivity analysis described here illustrates the point that our uncertainty regarding the life history of *Delma impar* should be translated with caution when interpreting the apparently optimistic results for the rural populations modeled herein. While urban populations do in fact seem to show less stability and greater extinction risk, the precise quantitative nature of the results are in large part determined by the particular set of demographic parameters that make up the individual modelling scenarios.

Finally, it is instructive to explore further the relative contributions made by increased juvenile mortality and wildfires to the destabilisation of urban populations of *D. impar*. This was done by repeating the urban population simulations with juvenile mortality set at 50% instead of the 60% value taken to be characteristic of these areas beset by higher predation rates. These results are shown in Tables 5.12 and 5.13. Of particular importance are the areas free from controlled burn management, as these are subjected to both the urban wildfire regime and the differential mortalities making up this analysis. Small populations with reduced juvenile mortality (i.e. Files 615 and 616) show a nearly 50% reduction in the rate of population decline compared to the corresponding scenarios with high juvenile mortality (Files 415 and 416), but the risk of extinction remains quite high at over 80%. Large populations (i.e., Files 619 and 620) show a dramatic decline in the risk of extinction within 100 years, although the rates of population growth remain negative but less severe than in the corresponding scenarios with higher juvenile mortality (Files 419 and 420). A very similar pattern emerges in scenarios where the populations are assumed to be at carrying capacity at the beginning of the simulations. It appears from this analysis that both higher juvenile mortality and the severe effects of yearly urban wildfires, despite their perceived relatively small effect, contribute significantly to the extinction threats faced by urban populations of *Delma impar*.

Table 5.6: Rural Populations / $N_0 \neq K$ / 1st year mortality = 50%

Filename	Modelling conditions	r	SD(r)	Prob. Extinct.	Mean final pop. Size (N)	SD(N)	Mean time to Extinc
DELMA401.OUT	small, rural, grazed	0.043	0.206	0.010	379.80	123.73	90.0
DELMA402.OUT	small, rural, grazed, no catastrophes	0.089	0.126	0.000	500.18	50.57	0.0
DELMA403.OUT	large, rural, grazed	0.043	0.201	0.000	3519.36	1223.39	0.0
DELMA404.OUT	large, rural, grazed, no catastrophes	0.090	0.123	0.000	4388.75	277.74	0.0
DELMA405.OUT	small, rural, non-grazed, controlled burns	0.029	0.139	0.000	419.53	121.62	0.0
DELMA406.OUT	small, rural, non-grazed, controlled burns, no catastrophes	0.050	0.111	0.000	501.49	34.40	0.0
DELMA407.OUT	small, rural, non grazed, no burns	0.039	0.188	0.010	398.13	135.95	79.0
DELMA408.OUT	small, rural, non-grazed, no burns, no catastrophes	0.087	0.094	0.000	516.48	12.50	0.0
DELMA409.OUT	large, rural, non-grazed, controlled burns	0.029	0.132	0.000	3897.09	839.03	0.0
DELMA410.OUT	large, rural, non-grazed, controlled burns, no catastrophes	0.049	0.106	0.000	4334.05	262.33	0.0
DELMA411.OUT	large, rural, non-grazed, no burns	0.043	0.179	0.000	3694.51	1071.38	0.0
DELMA412.OUT	large, rural, non-grazed, no burns, no catastrophes	0.088	0.090	0.000	4512.87	89.54	0.0

Table 5.7: Urban Populations / $N_0 \neq K$ / 1st year mortality = 60%

Filename	Modelling conditions	r	SD(r)	Prob. Extinct.	Mean final pop. Size (N)	SD(N)	Mean time to Extinc
DELMA413.OUT	small, urban, controlled burns	-0.019	0.175	0.350	71.42	73.16	72.6
DELMA414.OUT	small, urban, controlled burns, no catastrophes	0.000	0.138	0.080	191.61	145.12	77.4
DELMA415.OUT	small, urban, no burns	-0.127	0.235	1.000	0.00	0.00	31.1
DELMA416.OUT	small, urban, no burns, no catastrophes	-0.103	0.219	1.000	0.00	0.00	38.2
DELMA417.OUT	large, urban, controlled burns	-0.013	0.138	0.010	877.38	1044.37	92.0
DELMA418.OUT	large, urban, controlled burns, no catastrophes	0.007	0.117	0.000	2789.69	1248.11	0.0
DELMA419.OUT	large, urban, no burns	-0.109	0.185	0.980	5.00	1.41	61.0
DELMA420.OUT	large, urban, no burns, no catastrophes	-0.090	0.172	0.950	6.80	3.11	72.8

Table 5.8: Rural Populations / $N_0 = K$ / 1st year mortality = 50%

Filename	Modelling condition	r	SD(r)	Prob. Extinct.	Mean final pop. Size (N)	SD(N)	Mean time to Extinct
DELMA421.OUT	small, rural, grazed	0.044	0.211	0.030	75.95	27.79	52.3
DELMA422.OUT	small, rural, grazed, no catastrophes	0.086	0.135	0.000	92.52	10.45	0.0
DELMA423.OUT	large, rural, grazed	0.045	0.199	0.000	1411.74	383.45	0.0
DELMA424.OUT	large, rural, grazed, no catastrophes	0.090	0.123	0.000	1689.65	144.79	0.0
DELMA425.OUT	small, rural, non-grazed, controlled burns	0.027	0.152	0.020	83.80	16.85	83.0
DELMA426.OUT	small, rural, non-grazed, controlled burns, no catastrophes	0.049	0.124	0.000	91.08	12.78	0.0
DELMA427.OUT	small, rural, non grazed, no burns	0.035	0.199	0.030	76.18	28.01	70.3
DELMA428.OUT	small, rural, non-grazed, no burns, no catastrophes	0.085	0.106	0.000	98.66	4.61	0.0
DELMA429.OUT	large, rural, non-grazed, controlled burns	0.029	0.133	0.000	1486.34	280.07	0.0
DELMA430.OUT	large, rural, non-grazed, controlled burns, no catastrophes	0.050	0.105	0.000	1660.22	132.90	0.0
DELMA431.OUT	large, rural, non-grazed, no burns	0.040	0.183	0.000	1405.63	424.62	0.0
DELMA432.OUT	large, rural, non-grazed, no burns, no catastrophes	0.088	0.091	0.000	1737.23	43.30	0.0

Table 5.9: Urban Populations / $N_0 = K$ / 1st year mortality = 60%

Filename	Modelling condition	r	SD(r)	Prob. Extinct.	Mean final pop. Size (N)	SD(N)	Mean time to Extinct
DELMA433.OUT	small, urban, controlled burns	-0.022	0.184	0.440	38.27	25.29	66.6
DELMA434.OUT	small, urban, controlled burns, no catastrophes	0.001	0.147	0.130	56.53	28.87	72.7
DELMA435.OUT	small, urban, no burns	-0.119	0.233	1.000	0.00	0.00	32.6
DELMA436.OUT	small, urban, no burns, no catastrophes	-0.097	0.222	0.990	3.00	0.00	39.4
DELMA437.OUT	large, urban, controlled burns	-0.012	0.138	0.000	529.68	473.63	0.0
DELMA438.OUT	large, urban, controlled burns, no catastrophes	0.004	0.116	0.000	1101.50	472.60	0.0
DELMA439.OUT	large, urban, no burns	-0.106	0.190	0.990	4.00	0.00	63.9
DELMA440.OUT	large, urban, no burns, no catastrophes	-0.090	0.175	0.940	7.50	4.46	72.7

Table 5.10: Rural Populations / $N_0 \neq K$ / 1st year mortality = 60%

Filename	Modelling conditions	r	SD(r)	Prob. Extinct.	Mean final pop. Size (N)	SD(N)	Mean time to Extinct
DELMA501.OUT	small, rural, grazed	0.004	0.208	0.110	221.01	165.79	72.5
DELMA502.OUT	small, rural, grazed, no catastrophes	0.052	0.130	0.000	459.94	66.28	0.0
DELMA503.OUT	large, rural, grazed	0.007	0.197	0.000	2105.06	1446.12	0.0
DELMA504.OUT	large, rural, grazed, no catastrophes	0.054	0.125	0.000	4285.37	404.83	0.0
DELMA505.OUT	small, rural, non-grazed, controlled burns	-0.012	0.164	0.210	106.57	116.27	69.0
DELMA506.OUT	small, rural, non-grazed, controlled burns, no catastrophes	0.008	0.128	0.020	259.58	161.16	81.5
DELMA507.OUT	small, rural, non grazed, no burns	-0.001	0.199	0.170	200.12	154.62	62.4
DELMA508.OUT	small, rural, non-grazed, no burns, no catastrophes	0.048	0.106	0.000	497.72	31.40	0.0
DELMA509.OUT	large, rural, non-grazed, controlled burns	-0.010	0.135	0.000	943.34	845.91	0.0
DELMA510.OUT	large, rural, non-grazed, controlled burns, no catastrophes	0.011	0.115	0.000	2908.26	1231.36	0.0
DELMA511.OUT	large, rural, non-grazed, no burns	0.003	0.181	0.000	1929.82	1475.31	0.0
DELMA512.OUT	large, rural, non-grazed, no burns, no catastrophes	0.048	0.100	0.000	4318.66	328.87	0.0

Table 5.11: Rural Populations / $N_0 = K$ / 1st year mortality = 60% (continued over page)

Filename	Modelling Conditions	r	SD(r)	Prob. Extinct.	Mean final pop. Size (N)	SD(N)	Mean time to Extinct
DELMA521.OUT	small, rural, grazed	0.001	0.226	0.270	44.85	30.74	56.4
DELMA522.OUT	small, rural, grazed, no catastrophes	0.051	0.140	0.000	83.07	17.26	0.0
DELMA523.OUT	large, rural, grazed	0.005	0.201	0.020	753.71	534.15	73.5
DELMA524.OUT	large, rural, grazed, no catastrophes	0.052	0.126	0.000	1633.77	157.82	0.0
DELMA525.OUT	small, rural, non-grazed, controlled burns	-0.018	0.179	0.420	42.62	26.70	70.1
DELMA526.OUT	small, rural, non-grazed, controlled burns, no catastrophes	0.008	0.141	0.050	63.85	28.11	71.6
DELMA527.OUT	small, rural, non grazed, no burns	-0.013	0.223	0.470	46.57	31.84	68.5
DELMA528.OUT	small, rural, non-grazed, no burns, no catastrophes	0.046	0.117	0.000	94.15	8.28	0.0

DELMA529.OUT	large, rural, non-grazed, controlled burns	-0.009	0.138	0.000	612.47	469.46	0.0
DELMA530.OUT	large, rural, non-grazed, controlled burns, no catastrophes	0.012	0.116	0.000	1303.60	402.25	0.0
DELMA531.OUT	large, rural, non-grazed, no burns	0.002	0.184	0.000	822.83	571.91	0.0
DELMA532.OUT	large, rural, non-grazed, no burns, no catastrophes	0.052	0.101	0.000	1669.53	114.06	0.0

Table 5.12: Urban Populations / $N_0 \neq K$ / 1st year mortality = 50%

Filename	Modelling Conditions	r	SD(r)	Prob. Extinct.	Mean final pop. Size (N)	SD(N)	Mean time to Extinct
DELMA613.OUT	small, urban, controlled burns	0.029	0.139	0.010	425.19	120.13	88.0
DELMA614.OUT	small, urban, controlled burns, no catastrophes	0.049	0.111	0.000	491.94	44.92	0.0
DELMA615.OUT	small, urban, no burns	-0.074	0.217	0.990	2.00	0.00	52.6
DELMA616.OUT	small, urban, no burns, no catastrophes	-0.051	0.187	0.820	16.33	11.13	60.9
DELMA617.OUT	large, urban, controlled burns	0.030	0.133	0.000	3832.76	779.93	0.0
DELMA618.OUT	large, urban, controlled burns, no catastrophes	0.050	0.106	0.000	4338.11	328.49	0.0
DELMA619.OUT	large, urban, no burns	-0.066	0.164	0.550	17.11	19.68	83.0
DELMA620.OUT	large, urban, no burns, no catastrophes	-0.039	0.105	0.050	56.74	45.65	92.0

Table 5.13: Urban Populations / $N_0 = K$ / 1st year mortality = 50%

Filename	Modelling Conditions	r	SD(r)	Prob. Extinct.	Mean final pop. Size (N)	SD(N)	Mean time to Extinct
DELMA633.OUT	small, urban, controlled burns	0.025	0.152	0.010	74.00	23.73	95.0
DELMA634.OUT	small, urban, controlled burns, no catastrophes	0.046	0.124	0.000	91.43	14.78	0.0
DELMA635.OUT	small, urban, no burns	-0.071	0.209	0.940	12.00	5.97	47.7
DELMA636.OUT	small, urban, no burns, no catastrophes	-0.050	0.192	0.810	14.63	11.75	61.6
DELMA637.OUT	large, urban, controlled burns	0.026	0.133	0.000	1499.99	292.43	0.0
DELMA638.OUT	large, urban, controlled burns, no catastrophes	0.051	0.106	0.000	1655.32	132.45	0.0
DELMA639.OUT	large, urban, no burns	-0.064	0.161	0.530	18.77	17.52	84.4
DELMA640.OUT	large, urban, no burns, no catastrophes	-0.038	0.110	0.050	63.85	55.17	89.6

5.3 Influence of Fire Frequency

5.3.1 Model parameters

The frequency of fire events will be strongly influenced by habitat management practices such as controlled burning regimes. The impacts of various frequencies of burning on the viability of *D. impar* populations were modelled. The baseline population was subjected to varying frequencies of 'controlled' burning regimes, and wildfire events as described in Table 5.14.

Table 5.14 Potential fire regimes in *D. impar* habitat

Regime	Frequency	Effect on Reproduction	Effects on Survival
Wildfire 1 (frequent, low density) 10% habitat burnt	1/1yrs	0.9	0.9
Wildfire 2 (less frequent, greater severity) 80% habitat burnt	1/25yrs	0.2	0.6
Controlled Fire (20% habitat burnt)	1/1, 1/3, 1/10yrs	0.6	0.95

Very frequent (1/yr), low impact wildfires have been reported in ACT grassland areas in close proximity to urban areas; the high frequency is ascribed to human agency. Wildfires in rural grasslands are assumed to be less frequent (1/25yrs), and to have more severe immediate effects, burning on average 80% of habitat and reducing reproduction to 20% of normal output, while decreasing adult survivorship by 40%.

Controlled burning was assumed to affect 20% of an area, on average, and higher intensity burning has been assumed. A regime of controlled burning in the September - October period was modelled, with the resulting impact assumed to be much greater on reproductive output (reduced by 40%) than adult survivorship (reduced by 5%).

The effect of the differing wildfire frequencies was modelled, as was the effect of varying frequencies of controlled fire regimes.

All simulations also included a 10% probability of drought with sufficient severity to depress reproduction by 80% and reduce survivorship by 10%, as described in the baseline scenario.

5.3.2 Results

Annual events of Fire

In all cases where fire (wild or controlled) occurred at a frequency of once every year or greater, a negative population growth rate resulted, with a probability of the populations becoming extinct. Most cases of extinction were calculated to occur after a mean of 78 years. However, the most extreme case, where wildfire and controlled fire both occurred at yearly frequencies, indicated an 18% yearly population decline, with a mean time to extinction of 38.5 years.

Events of fire occurring at frequencies of once every three years or greater

In simulations for populations where controlled burning occurred at a frequency of once every three years or greater, positive growth rates (at a minimum of 5%) were predicted, and no probability of extinction was detected. Even when subjected to the severity of impact predicted for 'rural' wildfires, populations retained positive growth rates.

Table 5.15 The impact of fire frequency on the baseline population

File name	Condition	Growth rate		Prob. Extinction	Pop. Size		Mean time to extinct.
		r	SD(r)		N	SD(N)	
DELMA204.OUT	Wildfire 2 (severe; 1/25yrs)	0.033	0.232	0.000	2676.44	1182.06	0.0
DELMA205.OUT	Wildfire 1 (less severe; annual)	-0.028	0.134	0.020	227.99	253.19	94.5
DELMA206.OUT	Controlled burn (annual)	-0.049	0.133	0.190	38.99	36.91	88.2
DELMA207.OUT	Controlled burn (1/3yrs)	0.054	0.145	0.000	3643.49	586.44	0.0
DELMA208.OUT	Controlled burn (1/10yrs)	0.085	0.138	0.000	3849.98	296.18	0.0

r = mean instantaneous growth rate; SD(r) = standard deviation of r; N = mean final population size; SD(N) = standard deviation of N; Het = mean heterozygosity retained

5.4 The impact of predation on urban populations

5.4.1 Assessment of tail loss data

Rationale

Tail loss (autotomy) is considered to be a mechanism for surviving predation and is utilised by several reptilian families including *pygopodids*. Many trapped specimens of *D. impar* show evidence of tail loss /regrowth. The capture data were examined to determine if levels of tail loss/regrowth in animals from rural and urban areas differed, and therefore, whether they could be used to assess relative predation pressure.

The ACT and Victorian data were considered separately.

Victoria

For Victoria, the sites were considered in terms of whether they were close to Melbourne (Urban), or in country areas (Rural). In addition, one of the country sites was from South Australia (Bool Lagoon).

The animals caught which were found to have evidence of tail regrowth were expressed as a proportion of the total number of animals caught over one trapping period for each of the sites.

Table 5.16 Victorian Urban sites

Site	specimens (tail loss/total)	% tail loss	Total %
<i>Derrimut</i> (four trapping periods)	12/29	41%	
	0/27	0%	
	11/20	55%	
	2/19	11%	27%
<i>Albion</i>	26/67	39%	
	6/12	50%	44%
<i>Sunshine Tip</i>	5/14		36%
<i>Ravenhall (Deer Park)</i>	4/19		21%
Overall percentage for tail loss at urban sites			32%

Table 5.17 Victorian Rural sites

Site	specimens (tail loss/total)	% tail loss	Total %
<i>Cressy</i>	14/63		22%
<i>Bool Lagoon (S.A.)</i>	2/11		18%
<i>Rockbank</i>	0/14		0%
<i>Overall percentage for tail loss at rural sites</i>			13%

ACT

In the ACT six separate areas were considered in terms of relative percentage of tail loss. Data from about 900 individuals caught in the ACT were separated into distinct areas.

Table 5.18 ACT Urban sites

Site	specimens (tail loss/total)	% tail loss	Total %
<i>Gungahlin</i>			
Crace/Gungahderra	101/291	34%	
Town Centre (Mulangarri)	78/190	41%	
South Mitchell	26/71	36%	37%
<i>Kaleen</i>	17/31		54%
<i>Overall percentage for tail loss at urban sites</i>			46%

Table 5.19 ACT Urban sites

Site	specimens (tail loss/total)	% tail loss	Total %
<i>Jerabomberra Valley</i>	1/14		7%
<i>Majura Valley</i>	42/173		24%
<i>Yarramundi Reach</i> (National Museum)	4/16		25%
<i>Overall percentage for tail loss at rural sites</i>			19%

Urban/Rural boundary

One area in the ACT can be considered to be on the interface between urban and rural areas.

Kenny/Wells Station Road: 26/61 = 42%

Discussion

The results from both Victoria and the ACT indicate a higher percentage of tail loss in urban areas compared with sites

in rural areas, although this result has not been statistically verified. In addition, in the ACT, areas of urban/rural interface showed essentially the same proportion of tail loss as the urban areas.

It is not clear how this information correlates with types of predation nor actual mortality in this species, although it has been suggested that feral and domestic cats may be more abundant close to urban areas. A closely related species (*Delma inornata*) which occurs with *D. impar* at some sites in both Victoria and the ACT, has been recorded to be killed by roaming domestic cats. The presence of more perching sites (such as telegraph poles, etc.) in urban situations may increase the effectiveness of predation by raptors and this issue has been raised as a topic for further research in Victoria. In the Derrimut Grassland in Victoria, predation by some waterbird species is another potential cause of mortality, as there is a record of a White-faced Heron being lodged at the South Australian Museum which was found to contain 20 *Leiolopisma pagenstecheri*, a skink species sympatric with *Delma impar*. This finding has implications for *D. impar* in both rural and urban areas.

5.4.2 Population Structure

Purpose

To establish whether there is a difference in population structure in rural and urban areas, which may provide information on differences in recruitment and longevity of populations in environments with potential differences in predation pressure

Methods

All SVL for all sites for all years were combined for each of the ACT and Victorian data sets. These were plotted as histograms.

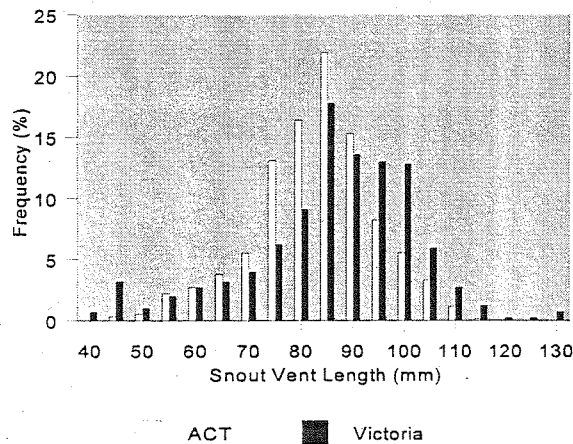
All SVL data for Victorian sites classed as “rural” (in country areas) were separated from those classified as “urban” (close to Melbourne), sorted into 5mm length classes and plotted on a histogram.

The SVL data set collected for all years from sites near Kaleen (a suburb of Canberra) and a similar data set from sites in the Majura Valley (an area supporting a nature reserve, rural leases, a firing range and Canberra Airport Beacon Facility) were separated, sorted into 5mm length classes and plotted on a histogram.

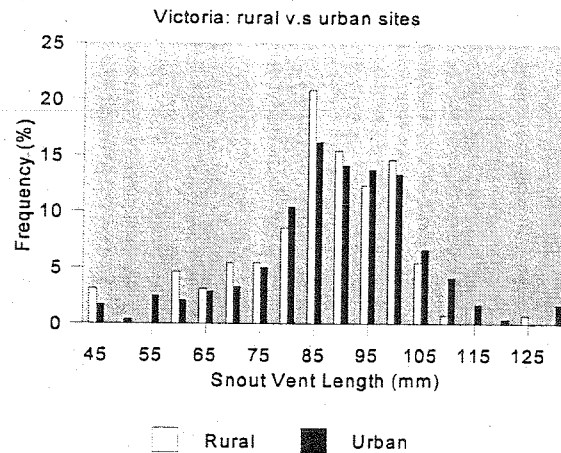
Results

Over 900 records of SVL have been made during several survey periods in the ACT since the 1988-89 summer, and 411 records from Victorian sites. When plotted, these data fitted a normal distribution (Fig. 5.4). In all cases, the most frequent size class was in the 80-85 mm category.

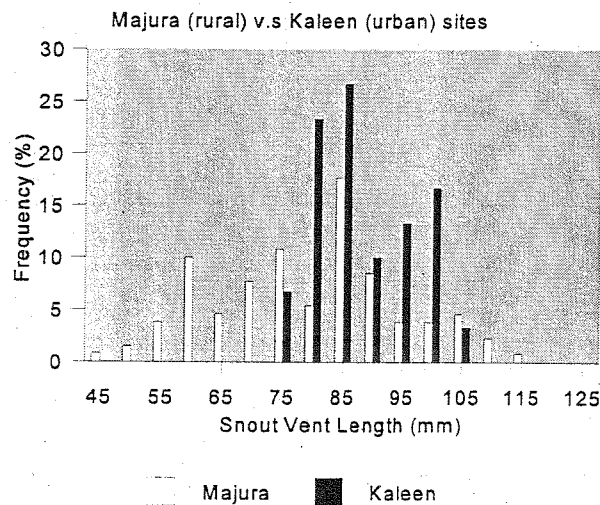
Fig 5.4. Size structure of *D. impar* populations in ACT and Victoria



Similar SVL distributions were obtained for both the rural and urban Victorian data (Figs 3 & 4).

Fig 5.5. Size structure of *D. impar* populations in rural and urban sites in Victoria

The two smaller subsets of ACT data demonstrated different trends. The Majura Valley data, which was based on 112 records, was skewed towards the smaller SVL classes. However, the Kaleen data, which was based on a smaller sample ($n = 30$), did not include any animals with a SVL of less than 70 mm (Fig 5.6).

Fig 5.6. Size structure of *D. impar* populations in rural and urban sites in ACT: Majura (rural) v.s Kaleen (urban) sites

Discussion

If equal catchability is assumed, the normal curve does not represent the expected SVL distribution of a population of *Delma impar*; a much higher proportion of young is expected. This could be explained by the period when most trapping for the species occurs, ie. November and December. During this period, adults appear to be most active and many gravid females are captured. Surveying in January and February would be expected to produce more hatchlings.

It is well known that the rate of growth in many reptiles is at least partly dependant on environmental conditions, rather than just age. Hence, in *D. impar* each SVL size class may not represent the same range in age classes. For example SVL size classes up to the 80-85 mm group may represent individuals aged from hatching to three years, whilst SVL size classes beyond that grouping may represent individuals aged from 4 to 20 years.

There was little apparent difference in the SVL data for rural areas and urban areas in Victoria. The data for the ACT demonstrated that there is variability in the SVL's between areas. However, different trapping regimes have been used

at each area over the years and more analysis is required before the results can be interpreted with any confidence. For example, the low number of small animals near the suburb of Kaleen may be an artefact of the small sample size, or it may reflect the structure of the population during November and December before hatching. Nevertheless, the data could show a population which is ageing, with no recruitment or with high predation of juveniles.

The Majura site, which is based on a larger sample size, contained more smaller animals. Further analysis of this data is required before any interpretation of possible predation pressure or juvenile mortality can be made. The higher proportion of small animals could be biased by data which has been collected outside the "normal" survey period of November and December.

Surveying in the ACT has been undertaken to determine presence of the species in proposed urban development areas. Consequently, surveying has been concentrated during the period when animals are most likely to be detected, ie. November and December when adults appear to be most active. Hence the surveys may well have been biased towards capture of adults. Now that extensive surveying has been completed in the ACT, surveying should focus on monitoring and sampling the population at other times of the year to obtain more information on other components of the population, especially hatchlings and juveniles. This will require refinement of trapping techniques to ensure all size classes are captured during surveys.

5.5 Recommendations

5.5.1 Research recommendations

More research is needed to elucidate longevity, age-specific mortalities, and female reproductive success; with age-specific mortality rates perhaps the most important to clarify. The following research recommendations are directed at further investigating these life history parameters.

1. Longevity research through close monitoring of captive animals; skeletal chronology, including using museum specimens; and trialing the use of tail tips for aging and genetic analysis.
2. Extend the wild population survey period, particularly targeting Jan - April, to provide data to better assess first year survivorship and proportion of juveniles in the population.
3. Refine survey techniques to ensure sampling across all age classes, and during extended trapping periods.
4. Determine types and rates of predation in rural and urban area.
5. Investigate the potential of tail-loss as a true indicator of relative mortality.
6. Determine differences in predation between juveniles and adults.
7. Investigate the effects of tail loss on reproduction and winter survivorship.
8. Determine success of estimates of gravidity through the use of direct investigative techniques, such as x-ray and ultra-sound to confirm visual assessments.
9. Determine reliable external characteristics for sex determination.
10. Investigate gonad development in museum specimens.
11. Achieve captive breeding; use captive population to investigate egg development and laying stages.
12. Investigate the impacts of fires on *D. impar* populations.

5.5.2 Recommended management strategies

Modelling using baseline data suggests that, where the impact of fire on reproduction and survival is of the magnitude assumed to result from a 20% burn, population viability is highly sensitive to the frequency of that event, in particular frequencies of less than once in every three years.

13. Given that the impacts of fire on *D. impar* populations are not well understood, fire regimes that burn more frequently than once every three years, and that burn more than 20% of a site, should be avoided until further investigation (therefore, any one patch in a *D. impar* site should not be burnt more frequently than once in every 15 years).

Baseline Data Input File

```

DELMA101.OUT      ***Output Filename***
Y      ***Graphing Files?***
N      ***Each Iteration?***
Y      ***Screen display of graphs?***
100    ***Simulations***
100    ***Years***
10     ***Reporting Interval***
1      ***Populations***
N      ***Inbreeding Depression?***
Y      ***EV correlation?***
2      ***Types Of Catastrophes***
P      ***Monogamous, Polygynous, or Hermaphroditic***
3      ***Female Breeding Age***
2      ***Male Breeding Age***
10     ***Maximum Age***
0.500000 ***Sex Ratio***
2      ***Maximum Litter Size***
N      ***Density Dependent Breeding?***
0.000000 ***Population 1: Percent Litter Size 0***
0.000000 ***Population 1: Percent Litter Size 1***
100.000000 ***Population 1: Percent Litter Size 2***
20.000000 ***EV--Reproduction***
50.000000 ***Female Mortality At Age 0***
10.000000 ***EV--FemaleMortality***
20.000000 ***Female Mortality At Age 1***
4.000000 ***EV--FemaleMortality***
10.000000 ***Female Mortality At Age 2***
2.000000 ***EV--FemaleMortality***
10.000000 ***Adult Female Mortality***
2.000000 ***EV--AdultFemaleMortality***
50.000000 ***Male Mortality At Age 0***
10.000000 ***EV--MaleMortality***
20.000000 ***Male Mortality At Age 1***
4.000000 ***EV--MaleMortality***
10.000000 ***Adult Male Mortality***
2.000000 ***EV--AdultMaleMortality***
4.000000 ***Probability Of Catastrophe 1***
0.200000 ***Severity--Reproduction***
0.500000 ***Severity--Survival***
10.000000 ***Probability Of Catastrophe 2***
0.200000 ***Severity--Reproduction***
0.900000 ***Severity--Survival***
Y      ***All Males Breeders?***
Y      ***Start At Stable Age Distribution?***
2000   ***Initial Population Size***
4000   ***K***
0.000000 ***EV--K***
N      ***Trend In K?***
N      ***Harvest?***
N      ***Supplement?***
N      ***AnotherSimulation?***
    
```

Baseline Data Output File

VORTEX -- simulation of genetic and demographic stochasticity

DELMA101.OUT

Wed Jul 31 07:52:31 1996

1 population(s) simulated for 100 years, 100 iterations

No inbreeding depression

First age of reproduction for females: 3 for males: 2

Age of senescence (death): 10

Sex ratio at birth (proportion males): 0.50000

Population 1:

Polygynous mating; all adult males in the breeding pool.

Reproduction is assumed to be density independent.

0.00 (EV = 20.00 SD) percent of adult females produce litters of size 0

0.00 percent of adult females produce litters of size 1

100.00 percent of adult females produce litters of size 2

50.00 (EV = 10.00 SD) percent mortality of females between ages 0 and 1

20.00 (EV = 4.00 SD) percent mortality of females between ages 1 and 2

10.00 (EV = 2.00 SD) percent mortality of females between ages 2 and 3

10.00 (EV = 2.00 SD) percent annual mortality of adult females (3<=age<=10)

50.00 (EV = 10.00 SD) percent mortality of males between ages 0 and 1

20.00 (EV = 4.00 SD) percent mortality of males between ages 1 and 2

10.00 (EV = 2.00 SD) percent annual mortality of adult males (2<=age<=10)

EVs may have been adjusted to closest values

possible for binomial distribution.

EV in reproduction and mortality will be correlated.

Frequency of type 1 catastrophes: 4.000 percent
with 0.200 multiplicative effect on reproduction
and 0.500 multiplicative effect on survival

Frequency of type 2 catastrophes: 10.000 percent
with 0.200 multiplicative effect on reproduction
and 0.900 multiplicative effect on survival

Initial size of Population 1:

(set to reflect stable age distribution)

Age	1	2	3	4	5	6	7	8	9	10	Total	
	237	171	138	112	91	73	59	49	38	32	1000	Males
	237	171	138	112	91	73	59	49	38	32	1000	Females

Carrying capacity = 4000 (EV = 0.00 SD)

Deterministic population growth rate (based on females, with assumptions of no limitation of mates, no density dependence, and no inbreeding depression):

r = 0.076 lambda = 1.078 R0 = 1.528

Generation time for: females = 5.62 males = 4.89

Stable age distribution:	Age class	females	males
	0	0.173	0.173
	1	0.078	0.078
	2	0.056	0.056
	3	0.045	0.045
	4	0.037	0.037
	5	0.030	0.030
	6	0.024	0.024
	7	0.019	0.019
	8	0.016	0.016
	9	0.013	0.013
	10	0.010	0.010

Ratio of adult (>= 2) males to adult (>= 3) females: 1.288

Population 1

Year 10

N[Extinct] = 0, P[E] = 0.000
 N[Surviving] = 100, P[S] = 1.000
 Population size = 3159.61 (100.77 SE, 1007.71 SD)
 Expected heterozygosity = 0.999 (0.000 SE, 0.000 SD)
 Observed heterozygosity = 1.000 (0.000 SE, 0.000 SD)
 Number of extant alleles = 1739.37 (41.96 SE, 419.58 SD)

Year 20

N[Extinct] = 0, P[E] = 0.000
 N[Surviving] = 100, P[S] = 1.000
 Population size = 3351.11 (93.69 SE, 936.91 SD)
 Expected heterozygosity = 0.998 (0.000 SE, 0.002 SD)
 Observed heterozygosity = 0.999 (0.000 SE, 0.001 SD)
 Number of extant alleles = 1148.10 (27.23 SE, 272.35 SD)

Year 30

N[Extinct] = 0, P[E] = 0.000
 N[Surviving] = 100, P[S] = 1.000
 Population size = 3235.37 (104.15 SE, 1041.46 SD)
 Expected heterozygosity = 0.998 (0.000 SE, 0.002 SD)
 Observed heterozygosity = 0.998 (0.000 SE, 0.001 SD)
 Number of extant alleles = 843.26 (19.75 SE, 197.52 SD)

Year 40

N[Extinct] = 0, P[E] = 0.000
 N[Surviving] = 100, P[S] = 1.000
 Population size = 3148.11 (101.74 SE, 1017.37 SD)
 Expected heterozygosity = 0.997 (0.000 SE, 0.002 SD)
 Observed heterozygosity = 0.998 (0.000 SE, 0.002 SD)
 Number of extant alleles = 665.12 (15.32 SE, 153.20 SD)

Year 50

N[Extinct] = 0, P[E] = 0.000
 N[Surviving] = 100, P[S] = 1.000
 Population size = 3320.83 (98.73 SE, 987.26 SD)
 Expected heterozygosity = 0.996 (0.000 SE, 0.003 SD)
 Observed heterozygosity = 0.997 (0.000 SE, 0.003 SD)
 Number of extant alleles = 550.17 (12.64 SE, 126.40 SD)

Year 60

N[Extinct] = 0, P[E] = 0.000
 N[Surviving] = 100, P[S] = 1.000
 Population size = 3202.85 (111.59 SE, 1115.89 SD)
 Expected heterozygosity = 0.996 (0.000 SE, 0.003 SD)
 Observed heterozygosity = 0.996 (0.000 SE, 0.003 SD)
 Number of extant alleles = 461.02 (11.39 SE, 113.92 SD)

Year 70

N[Extinct] = 0, P[E] = 0.000
 N[Surviving] = 100, P[S] = 1.000
 Population size = 3262.17 (104.63 SE, 1046.34 SD)
 Expected heterozygosity = 0.995 (0.000 SE, 0.004 SD)
 Observed heterozygosity = 0.995 (0.000 SE, 0.003 SD)
 Number of extant alleles = 400.84 (10.00 SE, 99.97 SD)

Year 80

N[Extinct] = 0, P[E] = 0.000
 N[Surviving] = 100, P[S] = 1.000
 Population size = 3328.42 (98.63 SE, 986.27 SD)
 Expected heterozygosity = 0.994 (0.000 SE, 0.004 SD)
 Observed heterozygosity = 0.994 (0.000 SE, 0.004 SD)
 Number of extant alleles = 356.02 (8.82 SE, 88.19 SD)

Year 90

N[Extinct] = 0, P[E] = 0.000
 N[Surviving] = 100, P[S] = 1.000
 Population size = 3273.06 (103.62 SE, 1036.15 SD)
 Expected heterozygosity = 0.993 (0.000 SE, 0.005 SD)
 Observed heterozygosity = 0.994 (0.000 SE, 0.005 SD)
 Number of extant alleles = 319.89 (7.98 SE, 79.82 SD)

Year 100

N[Extinct] = 0, P[E] = 0.000
 N[Surviving] = 100, P[S] = 1.000
 Population size = 3248.73 (108.97 SE, 1089.70 SD)
 Expected heterozygosity = 0.992 (0.001 SE, 0.005 SD)
 Observed heterozygosity = 0.993 (0.000 SE, 0.005 SD)
 Number of extant alleles = 289.36 (7.44 SE, 74.43 SD)

In 100 simulations of Population 1 for 100 years:
 0 went extinct and 100 survived.

This gives a probability of extinction of 0.0000 (0.0000 SE),
 or a probability of success of 1.0000 (0.0000 SE).

Mean final population for successful cases was 3248.73 (108.97 SE, 1089.70 SD)

Age 1	2	Adults	Total	
389.15		1230.02	1619.17	Males
388.46	278.06	963.04	1629.56	Females

Without harvest/supplementation, prior to carrying capacity truncation,
 mean growth rate (r) was 0.0624 (0.0022 SE, 0.2174 SD)

Final expected heterozygosity was 0.9924 (0.0005 SE, 0.0052 SD)
 Final observed heterozygosity was 0.9931 (0.0005 SE, 0.0048 SD)
 Final number of alleles was 289.36 (7.44 SE, 74.43 SD)



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Section 6 ~ Bibliography



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Section 7 ~ Contributed Papers



Draft research program prepared for internal discussion within ACT Parks & Conservation Service

Research programs for captive Delma impar.

1. Freeze branding

- I. Continue branding experiments with specific emphasis on technique, position of brand and legibility of the mark in relation to developing a numbering system.
- II. Develop an effective and reliable numbering system for use with freeze branding, which is independent and does not overlap with the old heat branding technique.
- III. Follow up work to confirm long-term success of any marking technique is essential in determining its scope for use in the field.

Priority: High

Time frame: develop and monitor over 1996 and hopefully use in field in Nov./Dec. 1996.

2. PIT tagging

- I. Examine the use of internal passive chips for the numbering of individuals. Speak with zoos etc. to determine if this method is feasible for an animal as small as *D. impar*.
- II. Calculate the cost of setting up and running a mark-recapture study using 'bar code' style individual recognition.
- III. If the method is thought to be physically possible and cost effective, then trial it with the captive animals.

Priority: Medium (more important if freeze branding proves inappropriate)

Time frame: should be looked into over the winter of 1996 and could be trialed on Tidbinbilla Nature Reserve (TNR) animals in the spring of 1996 if research shows it to be potentially useful.

3. Tracking

- I. Research the methods available for relocating released *D. impar* in the field, with the aim of determining animal movements and seasonal patterns in micro-habitat use.
- II. Trial any potential method on the captive animals to determine the:
 - a) ease of attaching/implanting the device,
 - b) degree to which the device affects the movement and behaviour of the animal,
 - c) range at which an animal can be located,
 - d) length of time over which animal can be relocated after release before new batteries are required or device is shed, and
 - e) costs associated with the different methods.

Priority: High

Time frame: research into possible methods should be conducted during the winter of 1996 for trial on captive animals in spring and then field use during the Nov./Dec. trapping period.

4. Captive breeding (WRU, TNR)

- I. Literature review on captive breeding of similar animals. Also discuss with authorities such as Chris Banks and others.
- II. Attempt to sex all of the captive animals at Tidbinbilla Nature Reserve, possibly through the use of x-rays.
- III. Establish tanks with variable proportions of males to females and ensure the use of seasonal variation in temperature and day length in order to stimulate breeding activity.
- IV. Examine the possibility of establishing an outdoor enclosure in order to better enable the social interactions which may be necessary for successful lizard breeding. This last option could be used as a fall-back position if the captive breeding is not successful in the tanks. Other information could be collected from animals kept in an outdoor enclosure, including the trialling of tracking techniques.

Priority: Medium

Time frame: sexing of TNR animals would be advantageous in any event, so this should be done at the first available opportunity (probably spring or early summer 1996). Once the sex of all individuals is known, decisions can be made about sex ratios in different tanks and animals can be moved. Coordination between work done at TNR and Melbourne Zoo is important in gaining the maximum amount of information. Because there is little effort involved in this project, it should be conducted this year despite its low priority.

5. Relocation/supplementation

- I. Look into potential sites for the introduction of *D. impar* where there is suitable habitat but no current population.
- II. Trial outdoor enclosures at these sites with the aim of determining whether the animals are capable of surviving in the wild after their time in captivity. Once animals appear to have become established, remove enclosure and allow natural dispersal.

Priority: Medium to high

Time frame: stage one should be researched during the winter of 1996 and stage two should be conducted as circumstances dictate.

Considerations

There should not be mixing between clusters unless determined to be necessary.

1. Data needs to be collected on the success of supplementation as well as reintroduction.

6. Examination of social behaviour and territoriality

- I. Determine the most effective method for observing the behaviour of the animals in enclosures (indoor and outdoor), probably through the use of a video camera.
- II. Vary the ratio of males to females and the density of animals in enclosures and observe the effect of this on behaviour.

Priority: Medium

Time frame: preferably this project would be conducted over the summer of 1996-97, but this is dependent on finding someone to do it. It would be even more valuable if it could be done at several different times of the year and over several years.

7. Trialling of new trapping techniques and attractants

- I. Communication between the involved groups in the different states is essential in conducting complementary work in different *D. impar* habitats over the geographic range to look at differential effectiveness of methods trialled.
- II. If any of this work is conducive to trialling on captive animals, then both the TNR and Melbourne Zoo animals may be available for trials.

Priority: High

Time frame: should be done whenever required and resources are available. Will require considerable human resources.

Techniques - Warwick Smith, ACT Parks & Conservation Service

FIELD TECHNIQUES

Analysis of data collected from surveys and field based research work has highlighted several important faults with current field techniques. These cause uncertainty in the data at all levels. As a result, the following recommendations have been made to direct research on improving techniques.

Development of improved trapping/monitoring techniques (Coulson, high)

This work must coincide with the development of tracking methods, as insights into the behaviour of the species may give clues as to what type of trap will be most effective. There are several ideas already developed which have had little or no field use.

The *Delma impar* working group in Victoria have just been given a grant to do this work. It may therefore be possible for efforts to have a different focus in the 1996 season in the ACT. If we are to conduct research into trapping techniques it should be complementing the work done in Victoria to avoid unnecessary duplication of effort. Further correspondence with the Victorian Working Group is required.

Current trapping methods are labour intensive and results from them are very difficult to interpret. An improved technique should exploit some aspect of the animal's behaviour which will maximise catch rates per unit effort. Examples include: providing artificial shelter sites where such shelter may be limiting; discovering some form of attractant whether it be shelter, food, reproductive or the promise of one of these. Hence, options to explore include:

- roof tiles or similar material
- artificial soil cracks
- sticky traps
- entanglement traps
- funnel traps
- low flat rooves over pits
- bait/vocalisation

TRACKING TECHNIQUES

There is a need to develop effective techniques for tracking animals over the medium and long-term, with the aim of determining:

- daily and seasonal movement patterns and activity periods, as well as age and sex-related differences in these patterns;
- area surveyed by pitfall traps with focus on determining population densities (Coulson, high)
- micro-habitat use;
- behavioural information to provide insights into the development of new trapping/monitoring methods (Coulson, high); and
- the success rate of reintroduction experiments (Coulson, medium);

The Victorian Working Group is trialling implantation of harmonic radar passive aerials in *D. inornata*. These trials will be conducted over the next few months and the results may provide an indication as to the usefulness of harmonic radar tracking for *D. impar*. If the technique has potential for *D. impar*, then it could be trialled initially on captive animals at Tidbinbilla Nature Reserve. The use of radio-tracking and fluoro-dye tracking should also be investigated and trialled if appropriate. The trial of these technique has the potential to harm or even kill individuals and should not be conducted during winter when animals are most susceptible to stress and disease. For this reason, the spring of 1996 will probably be the first available opportunity to trial selected techniques. If an effective method can be developed in the laboratory, then it should be trialled in the field at the earliest opportunity (probably during the 1996 Nov./Dec. trapping period).

Being able to track individuals would help answer questions relating to population size, mortality, home range, over-winter and overnight sites, daily and seasonal activity periods, and may also provide insights into improved trapping methods. Tracking animals this small is difficult with currently used methods. Their body form makes external attachment of tracking devices all but impossible and current technology is probably not small enough for internal implanting. Spooling has been attempted and has proven unsuccessful, and marking with fluorescent dye yielded limited data. Alternative techniques include: 1) implanting harmonic radar passive reflectors which will be trialled this season. However, *D. impar* may prove to be too small for implanting. 2) Marking of animals using radioactive isotopes may provide a very effective alternative. This technique can be used on animals of any size and can be used in dosages which will cause no increase in mortality or reproductive success. The use of radioactive isotopes has the potential to provide an enormous amount of information on many aspects of *D. impar* biology essential for conservation of the species. This would clearly require further research before its suitability for *D. impar* is determined. This technique has been widely used for beetles, amphibians, mammals and reptiles over many years.

MARKING TECHNIQUES

Investigation into an animal marking method which is more effective in the long term and more humane than heat branding

Heat branding animals is difficult, error prone, and causes stress and sometimes infection in the animal. An alternative technique which eliminates or minimises these problems is highly desirable. One alternative is the method often referred to as 'bar coding' or PIT tagging, but it is unsure at this time whether or not the chips are available in a small enough size as to not impede the movement of an animal as small and flexible as *D. impar*.

Work has already begun on freeze branding as an alternative to heat branding, but further branding and follow-up work is required before the technique can be adopted for field work. Other methods should be considered and trialled on captive populations in case freeze branding proves to be inappropriate. Trials of marking methods should be conducted before the winter of 1996 if possible, to allow the greatest possible time to assess effectiveness before field work in Nov./Dec.

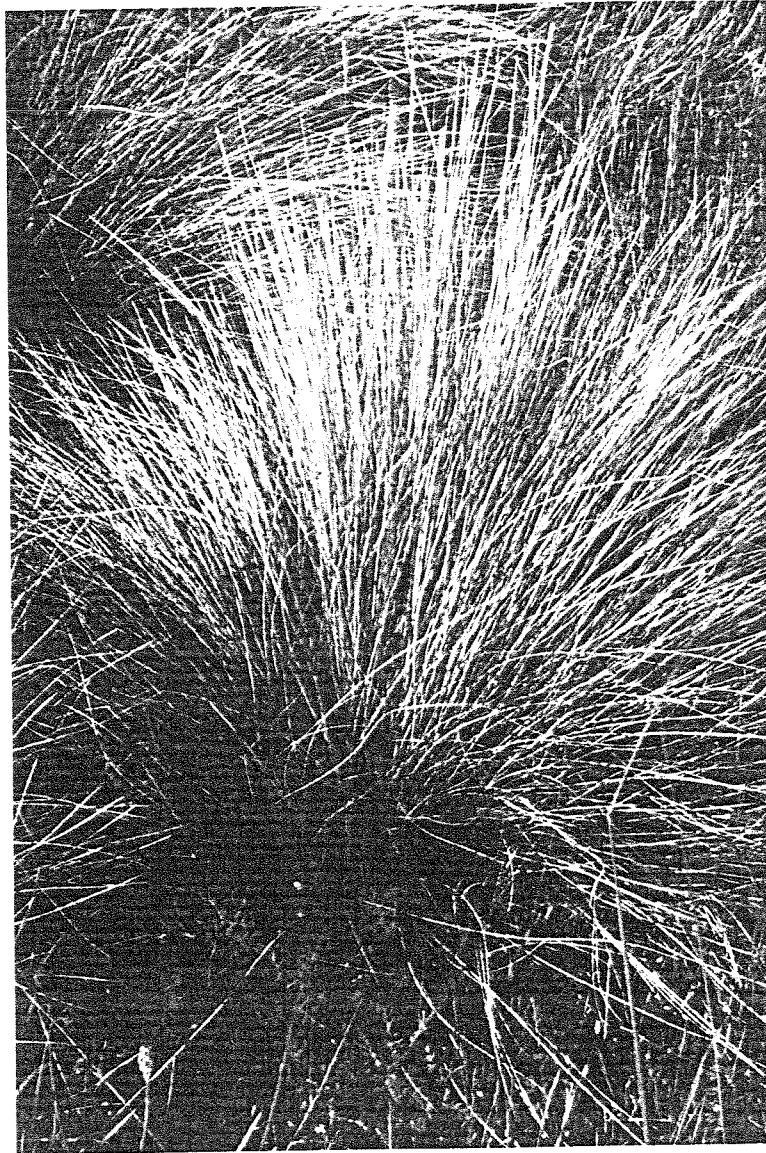
DELMA IMPAR
Striped Legless Lizard

Population & Habitat Viability Assessment Workshop

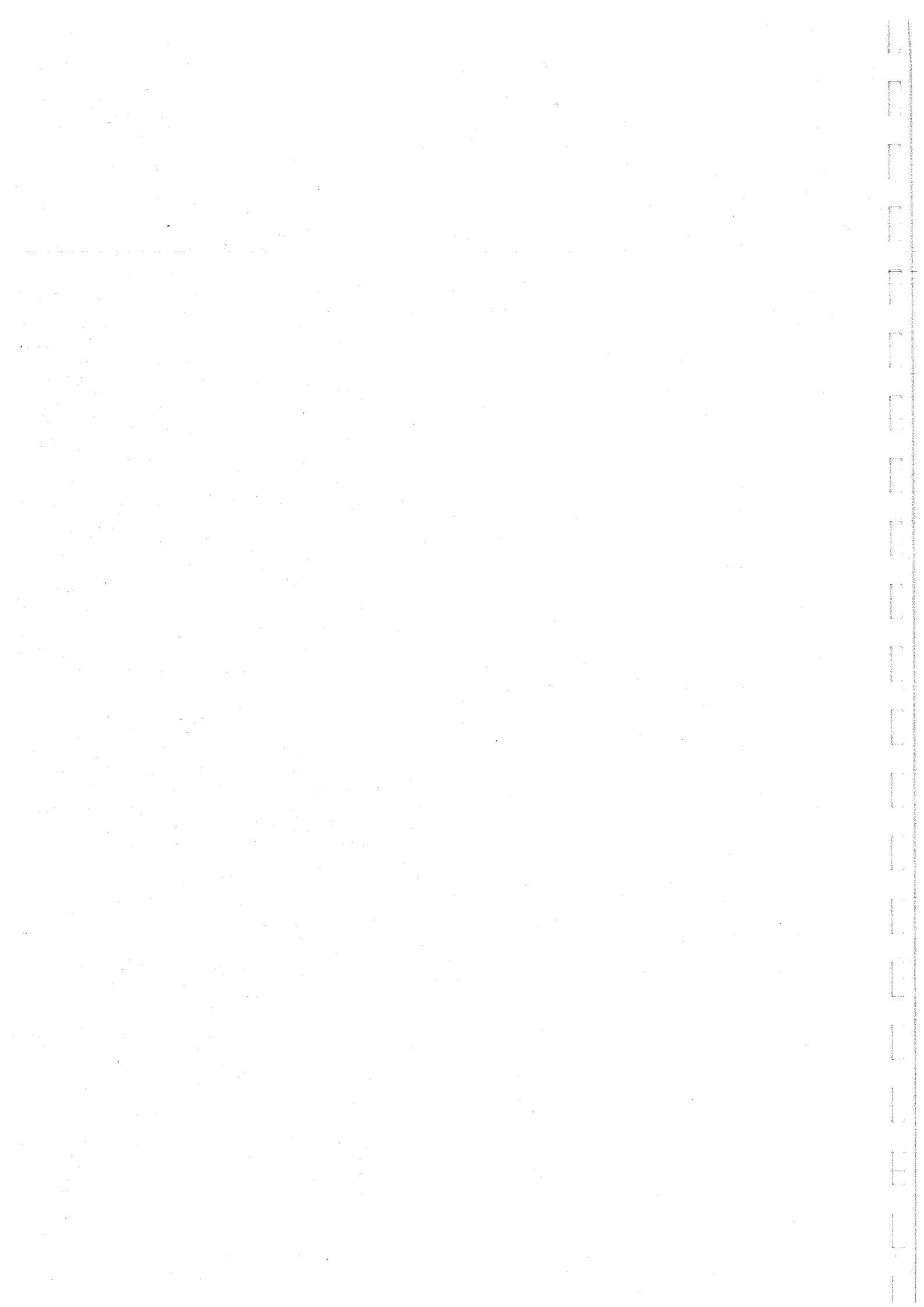
Canberra

30 July - 2 August 1996

FINAL REPORT



Section 8 ~ IUCN Policy Papers



Guidelines for re-introductions (as approved by 41st Meeting of Council, May 1995)

Final report • page 59

IUCN - the World Conservation Union
Re-introduction Specialist Group / Species Survival Commission

Introduction

These policy guidelines have been drafted by the Re-introduction Specialist Group of the IUCN's Species Survival Commission¹, in response to the increasing occurrence of re-introduction projects worldwide, and consequently, to the growing need for specific policy guidelines to help ensure that the re-introductions achieve their intended conservation benefit, and do not cause adverse side-effects of greater impact. Although IUCN developed a Position Statement on the Translocation of Living Organisms in 1987, more detailed guidelines were felt to be essential in providing more comprehensive coverage of the various factors involved in re-introduction exercises.

These Guidelines are intended to act as a guide for procedures useful to re-introduction programmes and do not represent an inflexible code of conduct. Many of the points are more relevant to re-introductions using captive-bred individuals than to translocations of wild species. Others are especially relevant to globally endangered species with limited numbers of founders. Each re-introduction proposal should be rigorously reviewed on its individual merits. It should be noted that re-introduction is always a very lengthy, complex and expensive process.

Re-introductions or translocations of species for short-term, sporting or commercial purposes - where there is no intention to establish a viable population - are a different issue and beyond the scope of these guidelines. These include fishing and hunting activities.

This document has been written to encompass the full range of plant and animal taxa and is therefore general. It will be regularly revised. Handbooks for re-introducing individual groups of animals and plants will be developed in future.

Context

The increasing number of re-introductions and translocations led to the establishment of the IUCN Species Survival Commission's Re-introduction Specialist Group. A priority of the Group has been to update IUCN's 1987 Position Statement on the Translocation of Living Organisms, in consultation with IUCN's other Commissions.

It is important that the Guidelines are implemented in the context of IUCN's broader policies pertaining to biodiversity conservation and sustainable management of natural resources. The philosophy for environmental conservation and management of IUCN and other conservation bodies is stated in key documents such as "Caring for the Earth" and the "Global Biodiversity Strategy," which cover the broad themes of the need for approaches with community involvement and participation in sustainable natural resource conservation, an overall enhanced quality of human life and the need to conserve and, where necessary, restore ecosystems. With regard to the latter, the re-introduction of a species is one specific instance of restoration where, in general, only that species is missing. Full restoration of an array of plant and animal species has rarely been tried to date.

Restoration of single species of plants and animals is becoming more frequent around the world. Some succeed, many fail. As this form of ecological management is increasingly common, it has been a priority for the Species Survival Commission's Re-introduction Specialist Group to develop guidelines so that re-introductions are both justifiable and likely to succeed, and that the conservation world can learn from each initiative, whether

¹ Guidelines for determining procedures for disposal of species confiscated in trade are being developed separately by IUCN.

successful or not. It is hoped that these Guidelines, based on extensive review of case-histories and wide consultation across a range of disciplines will introduce more rigour into the concepts, design, feasibility and implementation of re-introduction despite the wide diversity of species and conditions involved.

Thus, the priority has been to develop guidelines that are of direct, practical assistance to those planning, approving or carrying out re-introductions. The primary audience of these Guidelines is, therefore, the practitioners (usually managers or scientists), rather than decision-makers in governments. Guidelines directed towards the latter group would inevitably have to go into greater depth on legal and policy issues.

1. Definition of terms

- i. "Re-introduction": an attempt to establish a species² in an area which was once part of its historical range, but from which it has been extirpated or become extinct³. ("Re-establishment" is a synonym, but implies that the re-introduction has been successful).
- ii. "Translocation": deliberate and mediated movement of wild individuals to an existing population of conspecifics.
- iii. "Re-enforcement/Supplementation": addition of individuals to an existing population of conspecifics.
- iv. "Conservation/Benign Introductions": an attempt to establish a species, for the purpose of conservation, outside its recorded distribution but within an appropriate habitat and eco-geographical area. This is a feasible conservation tool only when there is no remaining area left within a species' historic range.

2. Aims and objectives of re-introduction

- i. Aims: The principal aim of any re-introduction should be to establish a viable, free-ranging population in the wild, of a species, subspecies or race, which has become globally or locally extinct, or extirpated, in the wild. It should be re-introduced within the species' former natural habitat and range and should require minimal long-term management.
- ii. Objectives: The objectives of a re-introduction may include: to enhance the long-term survival of a species; to re-establish a keystone species (in the ecological or cultural sense) in an ecosystem; to maintain and/or restore natural biodiversity; to provide long-term economic benefits to the local and/or national economy; to promote conservation awareness, or a combination of these.

3. Multidisciplinary approach

A re-introduction requires a multidisciplinary approach involving a team of persons drawn from a variety of backgrounds. As well as government personnel, they may include persons from governmental natural resource management agencies, non-governmental Organisations, funding bodies, universities, veterinary institutions, zoos (and private animal breeders) and/or botanic gardens, with a full range of suitable expertise. Team leaders should be responsible for coordination between the various bodies and provision should be made for publicity and public education about the project.

² The taxonomic unit referred to throughout the document is species; it may be a lower taxonomic unit (e.g. Sub-species or race) as long as it can be unambiguously defined.

³ A taxon is Extinct when there is no reasonable doubt that the last individual has died.

4. Pre-project activities

4 a. Biological

i. Feasibility study and background research

- An assessment should be made of the taxonomic status of individuals to be re-introduced. They should preferably be of the same subspecies or race as those which were extirpated, unless adequate numbers are not available. An investigation of historical information about the loss and fate of individuals from the re-introduction area, as well as molecular genetic studies, should be undertaken in case of doubt as to individuals' taxonomic status. A study of genetic variation within and between populations of this and related taxa can also be helpful. Special care is needed when the population has long been extinct.
- Detailed studies should be made of the status and biology of wild populations (if they exist) to determine the species' critical needs. For animals, this would include descriptions of habitat preferences, intraspecific variation and adaptations to local ecological conditions, social behaviour, group composition, home range size, shelter and food requirements, foraging and feeding behaviour, predators and diseases. For migratory species, studies should include the potential migratory areas. For plants, it would include biotic and abiotic habitat requirements, dispersal mechanisms, reproductive biology, symbiotic relationships (e.g. with mycorrhizae, pollinators), insect pests and diseases. Overall, a firm knowledge of the natural history of the species in question is crucial to the entire re-introduction scheme.
- The species, if any, that has filled the void created by the loss of the species concerned, should be determined; an understanding of the effect the re-introduced species will have on the ecosystem is important for ascertaining the success of the re-introduced population.
- The build-up of the released population should be modelled under various sets of conditions, in order to specify the optimal number and composition of individuals to be released per year and the numbers of years necessary to promote establishment of a viable population.
- A Population and Habitat Viability Analysis will aid in identifying significant environmental and population variables and assessing their potential interactions, which would guide long-term population management.

ii. Previous Re-introductions

- Thorough research into previous re-introductions of the same or similar species and wide-ranging contacts with persons having relevant expertise should be conducted prior to and while developing the re-introduction protocol.

iii. Choice of release site and type

- The site should be within the historic range of the species. For an initial re-enforcement there should be few remnant wild individuals. For a re-introduction, there should be no remnant population to prevent disease spread, social disruption and introduction of alien genes. In some circumstances, a re-introduction or re-enforcement may have to be made into an area which is fenced or otherwise delimited, but it should be within the species' former natural habitat and range.
- A conservation/benign introduction should be undertaken only as a last resort when no opportunities for re-introduction into the original site or range exist and only when a significant contribution to the conservation of the species will result.

The re-introduction area should have assured, long-term protection (whether formal or otherwise).

iv. Evaluation of re-introduction site

- Availability of suitable habitat: re-introductions should only take place where the habitat and landscape requirements of the species are satisfied, and likely to be sustained for the foreseeable future. The possibility of natural habitat change since extirpation must be considered. Likewise, a change in the legal/political or cultural environment since the species' extirpation needs to be ascertained and evaluated as a possible constraint. The area should have sufficient carrying capacity to sustain growth of the re-introduced population and support a viable (self-sustaining) population in the long run.
- Identification and elimination, or reduction to a sufficient level, of previous causes of decline: could include disease; over-hunting; over-collection; pollution; poisoning; competition with or predation by introduced species; habitat loss; adverse effects of earlier research or management programmes; competition with domestic livestock, which may be seasonal.
- Where the release site has undergone substantial degradation caused by human activity, a habitat restoration programme should be initiated before the re-introduction is carried out.

v. Availability of suitable release stock

- It is desirable that source animals come from wild populations. If there is a choice of wild populations to supply founder stock for translocation, the source population should ideally be closely related genetically to the original native stock and show similar ecological characteristics (morphology, physiology, behaviour, habitat preference) to the original sub-population.
- Removal of individuals for re-introduction must not endanger the captive stock population or the wild source population. Stock must be guaranteed available on a regular and predictable basis, meeting specifications of the project protocol.
- Individuals should only be removed from a wild population after the effects of translocation on the donor population have been assessed, and after it is guaranteed that these effects will not be negative.
- If captive or artificially propagated stock is to be used, it must be from a population which has been soundly managed both demographically and genetically, according to the principles of contemporary conservation biology.
- Re-introductions should not be carried out merely because captive stocks exist, nor solely as a means of disposing of surplus stock.
- Prospective release stock, including stock that is a gift between governments, must be subjected to a thorough veterinary screening process before shipment from original source. Any animals found to be infected or which test positive for non-endemic or contagious pathogens with a potential impact on population levels, must be removed from the consignment, and the uninfected, negative remainder must be placed in strict quarantine for a suitable period before retest. If clear after retesting, the animals may be placed for shipment.
- Since infection with serious disease can be acquired during shipment, especially if this is intercontinental, great care must be taken to minimise this risk.
- Stock must meet all health regulations prescribed by the veterinary authorities of the recipient country and adequate provisions must be made for quarantine if necessary.

vi. Release of captive stock

- Most species of mammals and birds rely heavily on individual experience and learning as juveniles for their survival; they should be given the opportunity to acquire the necessary information to enable survival in the wild through training in their captive environment; a captive bred individual's

probability of survival should approximate that of a wild counterpart.

- Care should be taken to ensure that potentially dangerous captive-bred animals (such as large carnivores or primates) are not so confident in the presence of humans that they might be a danger to local inhabitants and/or their livestock.

4 b. Socio-economic and legal requirements

- Re-introductions are generally long-term projects that require the commitment of long-term financial and political support.
- Socio-economic studies should be made to assess impacts, costs and benefits of the re-introduction programme to local human populations.
- A thorough assessment of attitudes of local people to the proposed project is necessary to ensure long-term protection of the re-introduced population, especially if the cause of species' decline was due to human factors (e.g. over-hunting, over-collection, loss or alteration of habitat). The programme should be fully understood, accepted and supported by local communities.
- Where the security of the re-introduced population is at risk from human activities, measures should be taken to minimise these in the re-introduction area. If these measures are inadequate, the re-introduction should be abandoned or alternative release areas sought.
- The policy of the country to re-introductions and to the species concerned should be assessed. This might include checking existing provincial, national and international legislation and regulations, and provision of new measures and required permits as necessary.
- Re-introduction must take place with the full permission and involvement of all relevant government agencies of the recipient or host country. This is particularly important in re-introductions in border areas, or involving more than one state or when a re-introduced population can expand into other states, provinces or territories.
- If the species poses potential risk to life or property, these risks should be minimised and adequate provision made for compensation where necessary; where all other solutions fail, removal or destruction of the released individual should be considered. In the case of migratory/mobile species, provisions should be made for crossing of international/state boundaries.

5. Planning, preparation and release stages

- Approval of relevant government agencies and land owners, and coordination with national and international conservation organisations.
- Construction of a multidisciplinary team with access to expert technical advice for all phases of the programme.
- Identification of short- and long-term success indicators and prediction of programme duration, in the context of agreed aims and objectives.
- Securing adequate funding for all programme phases.
- Design of pre- and post-release monitoring programme so that each re-introduction is a carefully designed experiment, with the capability to test methodology with scientifically collected data. Monitoring the health of individuals, as well as the survival, is important; intervention may be necessary if the situation proves unforeseeably favourable.

- Appropriate health and genetic screening of release stock, including stock that is a gift between governments. Health screening of closely related species in the re-introduction area.
- If release stock is wild-caught, care must be taken to ensure that:
 - a) the stock is free from infectious or contagious pathogens and parasites before shipment; and
 - b) the stock will not be exposed to vectors of disease agents which may be present at the release site (and absent at the source site) and to which it may have no acquired immunity.
- If vaccination prior to release, against local endemic or epidemic diseases of wild stock or domestic livestock at the release site, is deemed appropriate, this must be carried out during the "Preparation Stage" so as to allow sufficient time for the development of the required immunity.
- Appropriate veterinary or horticultural measures as required to ensure health of released stock throughout the programme. This is to include adequate quarantine arrangements, especially where founder stock travels far or crosses international boundaries to the release site.
- Development of transport plans for delivery of stock to the country and site of re-introduction, with special emphasis on ways to minimise stress on the individuals during transport.
- Determination of release strategy (acclimatization of release stock to release area; behavioural training - including hunting and feeding; group composition, number, release patterns and techniques; timing).
- Establishment of policies on interventions (see below).
- Development of conservation education for long-term support; professional training of individuals involved in the long-term programme; public relations through the mass media and in local community; involvement where possible of local people in the programme.
- The welfare of animals for release is of paramount concern through all these stages.

6. Post-release activities

- Post-release monitoring is required of all (or a sample of) individuals. This most vital aspect may be by direct (e.g. tagging, telemetry) or indirect (e.g. spoor, informants) methods as suitable.
- Demographic, ecological and behavioural studies of released stock must be undertaken.
- Study of processes of long-term adaptation by individuals and the population.
- Collection and investigation of mortalities.
- Interventions (e.g. supplemental feeding; veterinary aid; horticultural aid) when necessary.
- Decisions for revision, rescheduling, or discontinuation of programme where necessary.
- Habitat protection or restoration to continue where necessary.
- Continuing public relations activities, including education and mass media coverage.
- Evaluation of cost-effectiveness and success of re-introduction techniques.
- Regular publication in scientific and popular literature.

THE IUCN POLICY STATEMENT ON CAPTIVE BREEDING

Prepared by the
SSC Captive Breeding Specialist Group

As approved by the 22nd Meeting of the IUCN Council Gland, Switzerland

4 September 1987

SUMMARY: Habitat protection alone is not sufficient if the expressed goal of the World Conservation Strategy the maintenance of biotic diversity, is to be achieved. Establishment of self-sustaining captive populations and other supportive intervention will be needed to avoid the loss of many species, especially those at high risk in greatly reduced, highly fragmented, and disturbed habitats captive breeding programmes need to be established before specks are reduced to critically low numbers, and thereafter need to be coordinated Internationally according to sound biological principles, with a view to the maintaining or re establishment of viable populations in the wild.

PROBLEM STATEMENT

IUCN data indicate that about 3 per cent of terrestrial earth is gazetted for protection. Some of this and much of the other 97 per cent is becoming untenable for many species, and remaining populations are being greatly reduced and fragmented. From modern population biology one can predict that many species will be lost under these conditions. On average more than one mammal, bird, or reptile species has been lost in each year this century. Since extinctions of most taxa outside these groups are not recorded, the loss rate for all species is much higher.

Certain groups of species are at particularly high risk, especially forms with restricted distribution, those of large body size, those of high economic value, those at the top of food chains, and those which occur only in climax habitats. Species in these categories are likely to be lost first, but a wide range of other forms are also at risk. Conservation over the long term will require management to reduce risk, including *ex situ* populations which could support and interact demographically and genetically with wild populations.

FEASIBILITY

Over 3,000 vertebrate species are being bred in zoos and other captive animal facilities. When a serious attempt is made, most species breed in captivity, and viable populations can be maintained over the long term. A wealth of experience is available in these institutions, including husbandry, veterinary medicine, reproductive biology, behaviour, and genetics. They offer space for supporting populations of many threatened taxa, using resources not competitive with those for *in situ* conservation. Such captive stocks have in the past provided critical support for some wild populations (e.g. American bison, *Bison bison*), and have been the sole escape from extinction for others which have since been re-introduced to the wild (e.g. Arabian oryx, *Oryx leucoryx*).

RECOMMENDATION

IUCN urges that those national and international organizations and those individual institutions concerned with maintaining wild animals in captivity commit themselves to a general policy of developing demographically self-sustaining captive populations of endangered species wherever necessary.

SUGGESTED PROTOCOL

WHAT: The specific problems of the species concerned need to be considered, and appropriate aims for a captive breeding programme made explicit.

WHEN: The vulnerability of small populations has been consistently underestimated. This has erroneously shifted the timing of establishment of captive populations to the last moment, when the crisis is enormous and when extinction is probable. Therefore, timely recognition of such situations is critical, and is dependent on information on wild population status, particularly that provided by the IUCN Conservation Monitoring Centre. Management to best reduce the risk of extinction requires the establishment of supporting captive populations much earlier, preferably when the wild population is still in the thousands. Vertebrate taxa with a current census below one thousand individuals in the wild require close and swift cooperation between field conservationists and captive breeding specialists, to make their effort complementary and minimize the likelihood of the extinction of these taxa.

HOW: Captive populations need to be founded and managed according to sound scientific principles for the primary purpose of securing the survival of species through stable, self-sustaining captive populations. Stable captive populations preserve the options of reintroduction and/or supplementation of wild populations.

A framework of international cooperation and coordination between captive-breeding institutions holding species at risk must be based upon agreement to cooperatively manage such species for demographic security and genetic diversity. The IUCN/SSC Captive Breeding Specialist Group is an appropriate advisory body concerning captive breeding science and resources.

Captive programmes involving species at risk should be conducted primarily for the benefit of the species and without commercial transactions. Acquisition of animals for such programmes should not encourage commercial ventures or trade. Whenever possible, captive programmes should be carried out in parallel with field studies and conservation efforts aimed at the species in its natural environment.

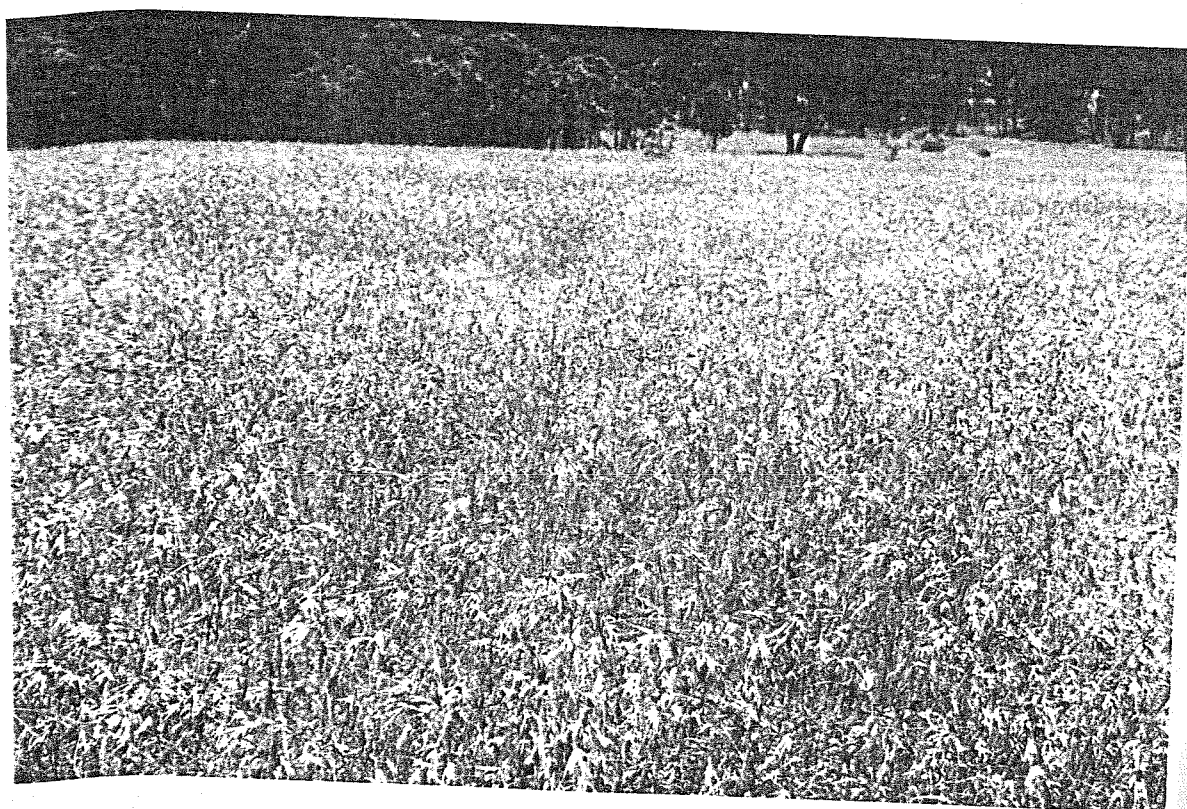
DELMA IMPAR
Striped Legless Lizard

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Section 9 ~ VORTEX Technical Reference

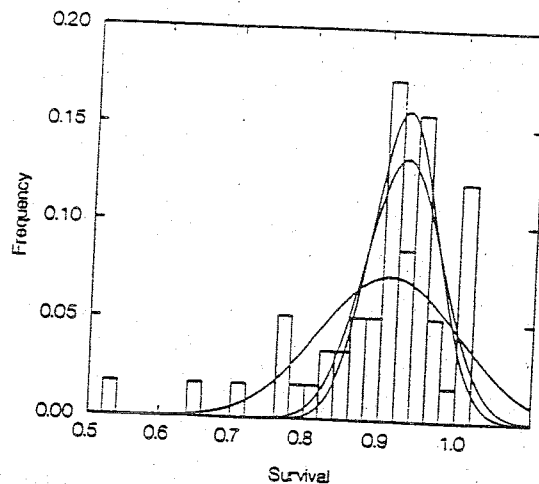


Fig. 1. Frequency histogram of the proportion of whooping cranes surviving each year, 1938-90. The broadest curve is the normal distribution that most closely fits the overall histogram. Statistically, this curve fits the data poorly. The second highest and second broadest curve is the normal distribution that most closely fits the histogram, excluding the five leftmost bars (7 outlier 'catastrophe' years). The narrowest and tallest curve is the normal approximation to the binomial distribution expected from demographic stochasticity. The difference between the tallest and second tallest curves is the variation in annual survival due to environmental variation.

hurricanes, large-scale fires, and floods are outliers in the distribution of environmental variation (e.g. five leftmost bars on Fig. 1). As a result, they have quantitatively and sometimes qualitatively different impacts on wildlife populations. (A forest fire is not just a very hot day.) Such events often precipitate the final decline to extinction (Simberloff 1986, 1988). For example, one of two populations of whooping crane was decimated by a hurricane in 1940 and soon after went extinct (Doughty 1989). The only remaining population of the black-footed ferret (*Mustela nigripes*) was being eliminated by an outbreak of distemper when the last 18 ferrets were captured (Clark 1989).

Genetic drift is the cumulative and non-adaptive fluctuation in allele frequencies resulting from the random sampling of genes in each generation. This can impede the recovery or accelerate the decline of wildlife populations for several reasons (Lacy 1993). Inbreeding, not strictly a component of genetic drift but correlated with it in small populations, has been documented to cause loss of fitness in a wide variety of species, including virtually all sexually reproducing animals in which the effects of inbreeding have been carefully studied (Wright 1977; Falconer 1981; O'Brien and Evermann 1988; Ralls *et al.* 1988; Lacy *et al.* 1993). Even if the immediate loss of fitness of inbred individuals is not large, the loss of genetic variation that results from genetic drift may reduce the ability of a population to adapt to future changes in the environment (Fisher 1958; Robertson 1960; Selander 1983).

Thus, the effects of genetic drift and consequent loss of genetic variation in individuals and populations have a negative impact on demographic rates and increase susceptibility to environmental perturbations and catastrophes. Reduced population growth and greater fluctuations in numbers in turn accelerate genetic drift (Crow and Kimura 1970). These synergistic destabilising effects of stochastic process on small populations of wildlife have been described as an 'extinction vortex' (Gilpin and Soulé 1986). The size below which a population is likely to be drawn into an extinction vortex can be considered a 'minimum

viable population' (MVP) (Seal and Lacy 1989), although Shaffer (1981) first defined a MVP more stringently as a population that has a 99% probability of persistence for 1000 years. The estimation of MVPs or, more generally, the investigation of the probability of extinction constitutes PVA (Gilpin and Soulé 1986; Gilpin 1989; Shaffer 1990).

Methods for Analysing Population Viability

An understanding of the multiple, interacting forces that contribute to extinction vortices is a prerequisite for the study of extinction-recolonisation dynamics in natural populations inhabiting patchy environments (Gilpin 1987), the management of small populations (Clark and Seebeck 1990), and the conservation of threatened wildlife (Shaffer 1981, 1990; Soulé 1987; Mace and Lande 1991). Because demographic and genetic processes in small populations are inherently unpredictable, the expected fates of wildlife populations can be described in terms of probability distributions of population size, time to extinction, and genetic variation. These distributions can be obtained in any of three ways: from analytical models, from empirical observation of the fates of populations of varying size, or from simulation models.

As the processes determining the dynamics of populations are multiple and complex, there are few analytical formulae for describing the probability distributions (e.g. Goodman 1987; Lande 1988; Burgmann and Gerard 1990). These models have incorporated only few of the threatening processes. No analytical model exists, for example, to describe the combined effect of demographic stochasticity and loss of genetic variation on the probability of population persistence.

A few studies of wildlife populations have provided empirical data on the relationship between population size and probability of extinction (e.g. Belovsky 1987; Berger 1990; Thomas 1990), but presently only order-of-magnitude estimates can be provided for MVPs of vertebrates (Shaffer 1987). Threatened species are, by their rarity, unavailable and inappropriate for the experimental manipulation of population sizes and long-term monitoring of undisturbed fates that would be necessary for precise empirical measurement of MVPs. Retrospective analyses will be possible in some cases, but the function relating extinction probability to population size will differ among species, localities and times (Lindenmayer *et al.* 1993).

Modelling the Dynamics of Small Populations

Because of the lack of adequate empirical data or theoretical and analytical models to allow prediction of the dynamics of populations of threatened species, various biologists have turned to Monte Carlo computer simulation techniques for PVA. By randomly sampling from defined probability distributions, computer programs can simulate the multiple, interacting events that occur during the lives of organisms and that cumulatively determine the fates of populations. The focus is on detailed and explicit modelling of the forces impinging on a given population, place, and time of interest, rather than on delineation of rules (which may not exist) that apply generally to most wildlife populations. Computer programs available to PVA include SPGPC (Grier 1980a, 1980b), GAPPs (Harris *et al.* 1986), RAMAS (Ferson and Akçakaya 1989; Akçakaya and Ferson 1990; Ferson 1990), FORPOP (Possingham *et al.* 1991), ALEX (Possingham *et al.* 1992), and SIMPOP (Lacy *et al.* 1989; Lacy and Clark 1990) and its descendant VORTEX.

SIMPOP was developed in 1989 by converting the algorithms of the program SPGPC (written by James W. Grier of North Dakota State University) from BASIC to the C programming language. SIMPOP was used first in a PVA workshop organised by the Species Survival Commission's Captive Breeding Specialist Group (IUCN), the United States Fish and Wildlife Service, and the Puerto Rico Department of Natural Resources to assist in planning and assessing recovery efforts for the Puerto Rican crested toad (*Peltophryne lemur*). SIMPOP was subsequently used in PVA modelling of other species threatened

with extinction, undergoing modification with each application to allow incorporation of additional threatening processes. The simulation program was renamed VORTEX (in reference to the extinction vortex) when the capability of modelling genetic processes was implemented in 1989. In 1990, a version allowing modelling of multiple populations was briefly named VORTICES. The only version still supported, with all capabilities of each previous version, is VORTEX Version 5.1.

VORTEX has been used in PVA to help guide conservation and management of many species, including the Puerto Rican parrot (*Amazona vittata*) (Lacy *et al.* 1989), the Javan rhinoceros (*Rhinoceros sondaicus*) (Seal and Foose 1989), the Florida panther (*Felis concolor coryi*) (Seal and Lacy 1989), the eastern barred bandicoot (*Perameles gunnii*) (Lacy and Clark 1990; Maguire *et al.* 1990), the lion tamarins (*Leontopithecus rosalia* ssp.) (Seal *et al.* 1990), the brush-tailed rock-wallaby (*Petrogale penicillata penicillata*) (Hill 1991), the mountain pygmy-possum (*Burramys parvus*), Leadbeater's possum (*Gymnobelideus leadbeateri*), the long-footed potoroo (*Potorous longipes*), the orange-bellied parrot (*Neophema chrysogaster*) and the helmeted honeyeater (*Lichenostomus melanops cassidix*) (Clark *et al.* 1991), the whooping crane (*Grus americana*) (Mirande *et al.* 1993), the Tana River crested mangabey (*Cercocebus galeritus galeritus*) and the Tana River red colobus (*Colobus badius rufomitratu*) (Seal *et al.* 1991), and the black rhinoceros (*Diceros bicornis*) (Foose *et al.* 1992). In some of these PVAs, modelling with VORTEX has made clear the insufficiency of past management plans to secure the future of the species, and alternative strategies were proposed, assessed and implemented. For example, the multiple threats to the Florida panther in its existing habitat were recognised as probably insurmountable, and a captive breeding effort has been initiated for the purpose of securing the gene pool and providing animals for release in areas of former habitat. PVA modelling with VORTEX has often identified a single threat to which a species is particularly vulnerable. The small but growing population of Puerto Rican parrots was assessed to be secure, except for the risk of population decimation by hurricane. Recommendations were made to make available secure shelter for captive parrots and to move some of the birds to a site distant from the wild flock, in order to minimise the damage that could occur in a catastrophic storm. These recommended actions were only partly implemented when, in late 1989, a hurricane killed many of the wild parrots. The remaining population of about 350 Tana River red colobus were determined by PVA to be so fragmented that demographic and genetic processes within the 10 subpopulations destabilised population dynamics. Creation of habitat corridors may be necessary to prevent extinction of the taxon. In some cases, PVA modelling has been reassuring to managers: analysis of black rhinos in Kenya indicated that many of the populations within sanctuaries were recovering steadily. Some could soon be used to provide animals for re-establishment or supplementation of populations previously eliminated by poaching. For some species, available data were insufficient to allow definitive PVA with VORTEX. In such cases, the attempt at PVA modelling has made apparent the need for more data on population trends and processes, thereby helping to justify and guide research efforts.

Description of VORTEX

Overview

The VORTEX computer simulation model is a Monte Carlo simulation of the effects of deterministic forces, as well as demographic, environmental and genetic stochastic events, on wildlife populations. VORTEX models population dynamics as discrete, sequential events that occur according to probabilities that are random variables, following user-specified distributions. The input parameters used by VORTEX are summarised in the first part of the sample output given in the Appendix.

VORTEX simulates a population by stepping through a series of events that describe an annual cycle of a typical sexually reproducing, diploid organism: mate selection,

reproduction, mortality, increment of age by one year, migration among populations, removals, supplementation, and then truncation (if necessary) to the carrying capacity. The program was designed to model long-lived species with low fecundity, such as mammals, birds and reptiles. Although it could and has been used in modelling highly fecund vertebrates and invertebrates, it is awkward to use in such cases as it requires complete specification of the percentage of females producing each possible clutch size. Moreover, computer memory limitations often hamper such analyses. Although VORTEX iterates life events on an annual cycle, a user could model 'years' that are other than 12 months' duration. The simulation of the population is itself iterated to reveal the distribution of fates that the population might experience.

Demographic Stochasticity

VORTEX models demographic stochasticity by determining the occurrence of probabilistic events such as reproduction, litter size, sex determination and death with a pseudo-random number generator. The probabilities of mortality and reproduction are sex-specific and pre-determined for each age class up to the age of breeding. It is assumed that reproduction and survival probabilities remain constant from the age of first breeding until a specified upper limit to age is reached. Sex ratio at birth is modelled with a user-specified constant probability of an offspring being male. For each life event, if the random value sampled from the uniform 0-1 distribution falls below the probability for that year, the event is deemed to have occurred, thereby simulating a binomial process.

The source code used to generate random numbers uniformly distributed between 0 and 1 was obtained from Maier (1991), according to the algorithm of Kirkpatrick and Stoll (1981). Random deviates from binomial distributions, with mean p and standard deviation s , are obtained by first determining the integral number of binomial trials, N , that would produce the value of s closest to the specified value, according to

$$N = p(1-p)/s^2.$$

N binomial trials are then simulated by sampling from the uniform 0-1 distribution to obtain the desired result, the frequency or proportion of successes. If the value of N determined for a desired binomial distribution is larger than 25, a normal approximation is used in place of the binomial distribution. This normal approximation must be truncated at 0 and at 1 to allow use in defining probabilities, although, with such large values of N , s is small relative to p and the truncation would be invoked only rarely. To avoid introducing bias with this truncation, the normal approximation to the binomial (when used) is truncated symmetrically around the mean. The algorithm for generating random numbers from a unit normal distribution follows Latour (1986).

VORTEX can model monogamous or polygamous mating systems. In a monogamous system, a relative scarcity of breeding males may limit reproduction by females. In polygamous or monogamous models, the user can specify the proportion of the adult males in the breeding pool. Males are randomly reassigned to the breeding pool each year of the simulation, and all males in the breeding pool have an equal chance of siring offspring.

The 'carrying capacity', or the upper limit for population size within a habitat, must be specified by the user. VORTEX imposes the carrying capacity via a probabilistic truncation whenever the population exceeds the carrying capacity. Each animal in the population has an equal probability of being removed by this truncation.

Environmental Variation

VORTEX can model annual fluctuations in birth and death rates and in carrying capacity as might result from environmental variation. To model environmental variation, each

demographic parameter is assigned a distribution with a mean and standard deviation that is specified by the user. Annual fluctuations in probabilities of reproduction and mortality are modelled as binomial distributions. Environmental variation in carrying capacity is modelled as a normal distribution. The variance across years in the frequencies of births and deaths resulting from the simulation model (and in real populations) will have two components: the demographic variation resulting from a binomial sampling around the mean for each year, and additional fluctuations due to environmental variation and catastrophes (see Fig. 1 and section on The Dynamics of Small Populations, above).

Data on annual variations in birth and death rates are important in determining the probability of extinction, as they influence population stability (Goodman 1987). Unfortunately, such field information is rarely available (but see Fig. 1). Sensitivity testing, the examination of a range of values when the precise value of a parameter is unknown, can help to identify whether the unknown parameter is important in the dynamics of a population.

Catastrophes

Catastrophes are modelled in VORTEX as random events that occur with specified probabilities. Any number of types of catastrophes can be modelled. A catastrophe will occur if a randomly generated number between zero and one is less than the probability of occurrence. Following a catastrophic event, the chances of survival and successful breeding for that simulated year are multiplied by severity factors. For example, forest fires might occur once in 50 years, on average, killing 25% of animals, and reducing breeding by survivors by 50% for the year. Such a catastrophe would be modelled as a random event with 0.02 probability of occurrence each year, and severity factors of 0.75 for survival and 0.50 for reproduction.

Genetic Processes

Genetic drift is modelled in VORTEX by simulation of the transmission of alleles at a hypothetical locus. At the beginning of the simulation, each animal is assigned two unique alleles. Each offspring is randomly assigned one of the alleles from each parent. Inbreeding depression is modelled as a loss of viability during the first year of inbred animals. The impacts of inbreeding are determined by using one of two models available within VORTEX: a Recessive Lethals model or a Heterosis model.

In the Recessive Lethals model, each founder starts with one unique recessive lethal allele and a unique, dominant non-lethal allele. This model approximates the effect of inbreeding if each individual in the starting population had one recessive lethal allele in its genome. The fact that the simulation program assumes that all the lethal alleles are at the same locus has a very minor impact on the probability that an individual will die because of homozygosity for one of the lethal alleles. In the model, homozygosity for different lethal alleles are mutually exclusive events, whereas in a multilocus model an individual could be homozygous for several lethal alleles simultaneously. By virtue of the death of individuals that are homozygous for lethal alleles, such alleles would be removed slowly by natural selection during the generations of a simulation. This reduces the genetic variation present in the population relative to the case with no inbreeding depression, but also diminishes the subsequent probability that inbred individuals will be homozygous for a lethal allele. This model gives an optimistic reflection of the impacts of inbreeding on many species, as the median number of lethal equivalents per diploid genome observed for mammalian populations is about three (Ralls *et al.* 1988).

The expression of fully recessive deleterious alleles in inbred organisms is not the only genetic mechanism that has been proposed as a cause of inbreeding depression. Some or

most of the effects of inbreeding may be a consequence of superior fitness of heterozygotes (heterozygote advantage or 'heterosis'). In the Heterosis model, all homozygotes have reduced fitness compared with heterozygotes. Juvenile survival is modelled according to the logarithmic model developed by Morton *et al.* (1956):

$$\ln S = A - BF$$

in which S is survival, F is the inbreeding coefficient, A is the logarithm of survival in the absence of inbreeding, and B is a measure of the rate at which survival decreases with inbreeding. B is termed the number of 'lethal equivalents' per haploid genome. The number of lethal equivalents per diploid genome, $2B$, estimates the number of lethal alleles per individual in the population if all deleterious effects of inbreeding were due to recessive lethal alleles. A population in which inbreeding depression is one lethal equivalent per diploid genome may have one recessive lethal allele per individual (as in the Recessive Lethals model, above), it may have two recessive alleles per individual, each of which confer a 50% decrease in survival, or it may have some other combination of recessive deleterious alleles that equate in effect with one lethal allele per individual. Unlike the situation with fully recessive deleterious alleles, natural selection does not remove deleterious alleles at heterotic loci because all alleles are deleterious when homozygous and beneficial when present in heterozygous combination with other alleles. Thus, under the Heterosis model, the impact of inbreeding on survival does not diminish during repeated generations of inbreeding.

Unfortunately, for relatively few species are data available to allow estimation of the effects of inbreeding, and the magnitude of these effects varies considerably among species (Falconer 1981; Ralls *et al.* 1988; Lacy *et al.* 1993). Moreover, whether a Recessive Lethals model or a Heterosis model better describes the underlying mechanism of inbreeding depression and therefore the response to repeated generations of inbreeding is not well-known (Brewer *et al.* 1990), and could be determined empirically only from breeding studies that span many generations. Even without detailed pedigree data from which to estimate the number of lethal equivalents in a population and the underlying nature of the genetic load (recessive alleles or heterosis), applications of PVA must make assumptions about the effects of inbreeding on the population being studied. In some cases, it might be considered appropriate to assume that an inadequately studied species would respond to inbreeding in accord with the median (3.14 lethal equivalents per diploid) reported in the survey by Ralls *et al.* (1988). In other cases, there might be reason to make more optimistic assumptions (perhaps the lower quartile, 0.90 lethal equivalents), or more pessimistic assumptions (perhaps the upper quartile, 5.62 lethal equivalents).

Deterministic Processes

VORTEX can incorporate several deterministic processes. Reproduction can be specified to be density-dependent. The function relating the proportion of adult females breeding each year to the total population size is modelled as a fourth-order polynomial, which can provide a close fit to most plausible density-dependence curves. Thus, either positive population responses to low-density or negative responses (e.g. Allee effects), or more complex relationships, can be modelled.

Populations can be supplemented or harvested for any number of years in each simulation. Harvest may be culling or removal of animals for translocation to another (unmodelled) population. The numbers of additions and removals are specified according to the age and sex of animals. Trends in the carrying capacity can also be modelled in VORTEX, specified as an annual percentage change. These changes are modelled as linear, rather than geometric, increases or decreases.

Migration among Populations

VORTEX can model up to 20 populations, with possibly distinct population parameters. Each pairwise migration rate is specified as the probability of an individual moving from one population to another. This probability is independent of the age and sex. Because of between-population migration and managed supplementation, populations can be recolonised. VORTEX tracks the dynamics of local extinctions and recolonisations through the simulation.

Output

VORTEX outputs (1) probability of extinction at specified intervals (e.g., every 10 years during a 100-year simulation), (2) median time to extinction if the population went extinct in at least 50% of the simulations, (3) mean time to extinction of those simulated populations that became extinct, and (4) mean size of, and genetic variation within, extant populations (see Appendix and Lindenmayer *et al.* 1993).

Standard deviations across simulations and standard errors of the mean are reported for population size and the measures of genetic variation. Under the assumption that extinction of independently replicated populations is a binomial process, the standard error of the probability of extinction (SE) is reported by VORTEX as

$$SE(p) = \sqrt{[p \times (1-p)/n]},$$

in which the frequency of extinction was p over n simulated populations. Demographic and genetic statistics are calculated and reported for each subpopulation and for the metapopulation.

Availability of the VORTEX Simulation Program

VORTEX Version 5.1 is written in the C programming language and compiled with the Lattice 80286C Development System (Lattice Inc.) for use on microcomputers using the MS-DOS (Microsoft Corp.) operating system. Copies of the compiled program and a manual for its use are available for nominal distribution costs from the Captive Breeding Specialist Group (Species Survival Commission, IUCN), 12101 Johnny Cake Ridge Road, Apple Valley, Minnesota 55124, U.S.A. The program has been tested by many workers, but cannot be guaranteed to be error-free. Each user retains responsibility for ensuring that the program does what is intended for each analysis.

Sequence of Program Flow

- (1) The seed for the random number generator is initialised with the number of seconds elapsed since the beginning of the 20th century.
- (2) The user is prompted for input and output devices, population parameters, duration of simulation, and number of iterations.
- (3) The maximum allowable population size (necessary for preventing memory overflow) is calculated as

$$N_{max} = (K + 3s) \times (1 + L)$$

in which K is the maximum carrying capacity (carrying capacity can be specified to change linearly for a number of years in a simulation, so the maximum carrying capacity can be greater than the initial carrying capacity), s is the annual environmental variation in the carrying capacity expressed as a standard deviation, and L is the specified maximum litter size. It is theoretically possible, but very unlikely, that a simulated population will exceed the calculated N_{max} . If this occurs then the program will give an error message and abort.

(4) Memory is allocated for data arrays. If insufficient memory is available for data arrays then N_{max} is adjusted downward to the size that can be accommodated within the available memory and a warning message is given. In this case it is possible that the analysis may have to be terminated because the simulated population exceeds N_{max} . Because N_{max} is often several-fold greater than the likely maximum population size in a simulation, a warning it has been adjusted downward because of limiting memory often will not hamper the analyses. Except for limitations imposed by the size of the computer memory (VORTEX can use extended memory, if available), the only limit to the size of the analysis is that no more than 20 populations exchanging migrants can be simulated.

(5) The expected mean growth rate of the population is calculated from mean birth and death rates that have been entered. Algorithms follow cohort life-table analyses (Ricklefs 1979). Generation time and the expected stable age distribution are also estimated. Life-table estimations assume no limitation by carrying capacity, no limitation of mates, and no loss of fitness due to inbreeding depression, and the estimated intrinsic growth rate assumes that the population is at the stable age distribution. The effects of catastrophes are incorporated into the life-table analysis by using birth and death rates that are weighted averages of the values in years with and without catastrophes, weighted by the probability of a catastrophe occurring or not occurring.

(6) Iterative simulation of the population proceeds via steps 7-26 below. For exploratory modelling, 100 iterations are usually sufficient to reveal gross trends among sets of simulations with different input parameters. For more precise examination of population behaviour under various scenarios, 1000 or more simulations should be used to minimise standard errors around mean results.

(7) The starting population is assigned an age and sex structure. The user can specify the exact age-sex structure of the starting population, or can specify an initial population size and request that the population be distributed according to the stable age distribution calculated from the life table. Individuals in the starting population are assumed to be unrelated. Thus, inbreeding can occur only in second and later generations.

(8) Two unique alleles at a hypothetical genetic locus are assigned to each individual in the starting population and to each individual supplemented to the population during the simulation. VORTEX therefore uses an infinite alleles model of genetic variation. The subsequent fate of genetic variation is tracked by reporting the number of extant alleles each year, the expected heterozygosity or gene diversity, and the observed heterozygosity. The expected heterozygosity, derived from the Hardy-Weinberg equilibrium, is given by

$$H_e = 1 - \sum(p_i^2),$$

in which p_i is the frequency of allele i in the population. The observed heterozygosity is simply the proportion of the individuals in the simulated population that are heterozygous. Because of the starting assumption of two unique alleles per founder, the initial population has an observed heterozygosity of 1.0 at the hypothetical locus and only inbred animals can become homozygous. Proportional loss of heterozygosity by means of random genetic drift is independent of the initial heterozygosity and allele frequencies of a population (assuming that the initial value was not zero) (Crow and Kimura 1970), so the expected heterozygosity remaining in a simulated population is a useful metric of genetic decay for comparison across scenarios and populations. The mean observed heterozygosity reported by VORTEX is the mean inbreeding coefficient of the population.

(9) The user specifies one of three options for modelling the effect of inbreeding: (a) no effect of inbreeding on fitness, that is, all alleles are selectively neutral, (b) each founder individual has one unique lethal and one unique non-lethal allele (Recessive Lethals option), or (c) first-year survival of each individual is exponentially related to its inbreeding coefficient (Heterosis option). The first case is clearly an optimistic one, as almost all diploid

populations studied intensively have shown deleterious effects of inbreeding on a variety of fitness components (Wright 1977; Falconer 1981). Each of the two models of inbreeding depression may also be optimistic, in that inbreeding is assumed to have an impact only on first-year survival. The Heterosis option allows, however, for the user to specify the severity of inbreeding depression on juvenile survival.

(10) Years are iterated via steps 11-25 below.

(11) The probabilities of females producing each possible litter size are adjusted to account for density dependence of reproduction (if any).

(12) Birth rate, survival rates and carrying capacity for the year are adjusted to model environmental variation. Environmental variation is assumed to follow binomial distributions for birth and death rates and a normal distribution for carrying capacity, with mean rates and standard deviations specified by the user. At the outset of each year a random number is drawn from the specified binomial distribution to determine the percentage of females producing litters. The distribution of litter sizes among those females that do breed is maintained constant. Another random number is drawn from a specified binomial distribution to model the environmental variation in mortality rates. If environmental variations in reproduction and mortality are chosen to be correlated, the random number used to specify mortality rates for the year is chosen to be the same percentile of its binomial distribution as was the number used to specify reproductive rate. Otherwise, a new random number is drawn to specify the deviation of age- and sex-specific mortality rates for their means. Environmental variation across years in mortality rates is always forced to be correlated among age and sex classes.

The carrying capacity (K) of the year is determined by first increasing or decreasing the carrying capacity at year 1 by an amount specified by the user to account for linear changes over time. Environmental variation in K is then imposed by drawing a random number from a normal distribution with the specified values for mean and standard deviation.

(13) Birth rates and survival rates for the year are adjusted to model any catastrophes determined to have occurred in that year.

(14) Breeding males are selected for the year. A male of breeding age is placed into the pool of potential breeders for that year if a random number drawn for that male is less than the proportion of breeding-age males specified to be breeding.

(15) For each female of breeding age, a mate is drawn at random from the pool of breeding males for that year. The size of the litter produced by that pair is determined by comparing the probabilities of each potential litter size (including litter size of 0, no breeding) to a randomly drawn number. The offspring are produced and assigned a sex by comparison of a random number to the specified sex ratio at birth. Offspring are assigned, at random, one allele at the hypothetical genetic locus from each parent.

(16) If the Heterosis option is chosen for modelling inbreeding depression, the genetic kinship of each new offspring to each other living animal in the population is determined. The kinship between a new animal, A , and another existing animal, B is

$$f_{AB} = 0.5 \times (f_{MB} + f_{PB})$$

in which f_{ij} is the kinship between animals i and j , M is the mother of A , and P is the father of A . The inbreeding coefficient of each animal is equal to the kinship between its parents, $F = f_{MP}$, and the kinship of an animal to itself is $f_{A,A} = 0.5 \times (1 + F)$. [See Ballou (1983) for a detailed description of this method for calculating inbreeding coefficients.]

(17) The survival of each animal is determined by comparing a random number to the survival probability for that animal. In the absence of inbreeding depression, the survival probability is given by the age and sex-specific survival rate for that year. If the Heterosis model of inbreeding depression is used and an individual is inbred, the survival probability is multiplied by e^{-bF} in which b is the number of lethal equivalents per haploid genome.

If the Recessive Lethals model is used, all offspring that are homozygous for a lethal allele are killed.

(18) The age of each animal is incremented by 1, and any animal exceeding the maximum age is killed.

(19) If more than one population is being modelled, migration among populations occurs stochastically with specified probabilities.

(20) If population harvest is to occur that year, the number of harvested individuals of each age and sex class are chosen at random from those available and removed. If the number to be removed do not exist for an age-sex class, VORTEX continues but reports that harvest was incomplete.

(21) Dead animals are removed from the computer memory to make space for future generations.

(22) If population supplementation is to occur in a particular year, new individuals of the specified age class are created. Each immigrant is assigned two unique alleles, one of which will be a recessive lethal in the Recessive Lethals model of inbreeding depression. Each immigrant is assumed to be genetically unrelated to all other individuals in the population.

(23) The population growth rate is calculated as the ratio of the population size in the current year to the previous year.

(24) If the population size (N) exceeds the carrying capacity (K) for that year, additional mortality is imposed across all age and sex classes. The probability of each animal dying during this carrying capacity truncation is set to $(N-K)/N$, so that the expected population size after the additional mortality is K .

(25) Summary statistics on population size and genetic variation are tallied and reported. A simulated population is determined to be extinct if one of the sexes has no representatives.

(26) Final population size and genetic variation are determined for the simulation.

(27) Summary statistics on population size, genetic variation, probability of extinction, and mean population growth rate, are calculated across iterations and printed out.

Assumptions Underpinning VORTEX

It is impossible to simulate the complete range of complex processes that can have an impact on wild populations. As a result there are necessarily a range of mathematical and biological assumptions that underpin any PVA program. Some of the more important assumptions in VORTEX include the following.

(1) Survival probabilities are density independent when population size is less than carrying capacity. Additional mortality imposed when the population exceeds K affects all age and sex classes equally.

(2) The relationship between changes in population size and genetic variability are examined for only one locus. Thus, potentially complex interactions between genes located on the same chromosome (linkage disequilibrium) are ignored. Such interactions are typically associated with genetic drift in very small populations, but it is unknown if, or how, they would affect population viability.

(3) All animals of reproductive age have an equal probability of breeding. This ignores the likelihood that some animals within a population may have a greater probability of breeding successfully, and breeding more often, than other individuals. If breeding is not at random among those in the breeding pool, then decay of genetic variation and inbreeding will occur more rapidly than in the model.

- (4) The life-history attributes of a population (birth, death, migration, harvesting, supplementation) are modelled as a sequence of discrete and therefore seasonal events. However, such events are often continuous through time and the model ignores the possibility that they may be aseasonal or only partly seasonal.
- (5) The genetic effects of inbreeding on a population are determined in VORTEX by using one of two possible models: the Recessive Lethals model and the Heterosis model. Both models have attributes likely to be typical of some populations, but these may vary within and between species (Brewer *et al.* 1990). Given this, it is probable that the impacts of inbreeding will fall between the effects of these two models. Inbreeding is assumed to depress only one component of fitness: first-year survival. Effects on reproduction could be incorporated into this component, but longer-term impacts such as increased disease susceptibility or decreased ability to adapt to environmental change are not modelled.
- (6) The probabilities of reproduction and mortality are constant from the age of first breeding until an animal reaches the maximum longevity. This assumes that animals continue to breed until they die.
- (7) A simulated catastrophe will have an effect on a population only in the year that the event occurs.
- (8) Migration rates among populations are independent of age and sex.
- (9) Complex, interspecies interactions are not modelled, except in that such community dynamics might contribute to random environmental variation in demographic parameters. For example, cyclical fluctuations caused by predator-prey interactions cannot be modelled by VORTEX.

Discussion

Uses and Abuses of Simulation Modelling for PVA

Computer simulation modelling is a tool that can allow crude estimation of the probability of population extinction, and the mean population size and amount of genetic diversity, from data on diverse interacting processes. These processes are too complex to be integrated intuitively and no analytic solutions presently, or are likely to soon, exist. PVA modelling focuses on the specifics of a population, considering the particular habitat, threats, trends, and time frame of interest, and can only be as good as the data and the assumptions input to the model (Lindenmayer *et al.* 1993). Some aspects of population dynamics are not modelled by VORTEX nor by any other program now available. In particular, models of single-species dynamics, such as VORTEX, are inappropriate for use on species whose fates are strongly determined by interactions with other species that are in turn undergoing complex (and perhaps synergistic) population dynamics. Moreover, VORTEX does not model many conceivable and perhaps important interactions among variables. For example, loss of habitat might cause secondary changes in reproduction, mortality, and migration rates, but ongoing trends in these parameters cannot be simulated with VORTEX. It is important to stress that PVA does not predict in general what will happen to a population; PVA forecasts the likely effects only of those factors incorporated into the model.

Yet, the use of even simplified computer models for PVA can provide more accurate predictions about population dynamics than the even more crude techniques available previously, such as calculation of expected population growth rates from life tables. For the purpose of estimating extinction probabilities, methods that assess only deterministic factors are almost certain to be inappropriate, because populations near extinction will commonly be so small that random processes dominate deterministic ones. The suggestion by Mace and Lande (1991) that population viability be assessed by the application of simple rules (e.g., a taxon be considered Endangered if the total effective population size is below 50 or the

total census size below 250) should be followed only if knowledge is insufficient to allow more accurate quantitative analysis. Moreover, such preliminary judgments, while often important in stimulating appropriate corrective measures, should signal, not obviate, the need for more extensive investigation and analysis of population processes, trends and threats.

Several good population simulation models are available for PVA. They differ in capabilities, assumptions and ease of application. The ease of application is related to the number of simplifying assumptions and inversely related to the flexibility and power of the model. It is unlikely that a single or even a few simulation models will be appropriate for all PVAs. The VORTEX program has some capabilities not found in many other population simulation programs, but is not as flexible as are some others (e.g., GAPPS; Harris *et al.* 1986). VORTEX is user-friendly and can be used by those with relatively little understanding of population biology and extinction processes, which is both an advantage and a disadvantage.

Testing Simulation Models

Because many population processes are stochastic, a PVA can never specify what will happen to a population. Rather, PVA can provide estimates of probability distributions describing possible fates of a population. The fate of a given population may happen to fall at the extreme tail of such a distribution even if the processes and probabilities are assessed precisely. Therefore, it will often be impossible to test empirically the accuracy of PVA results by monitoring of one or a few threatened populations of interest. Presumably, if a population followed a course that was well outside of the range of possibilities predicted by a model, that model could be rejected as inadequate. Often, however, the range of plausible fates generated by PVA is quite broad.

Simulation programs can be checked for internal consistency. For example, in the absence of inbreeding depression and other confounding effects, does the simulation model predict an average long-term growth rate similar to that determined from a life-table calculation? Beyond this, some confidence in the accuracy of a simulation model can be obtained by comparing observed fluctuations in population numbers to those generated by the model, thereby comparing a data set consisting of tens to hundreds of data points to the results of the model. For example, from 1938 to 1991, the wild population of whooping cranes had grown at a mean exponential rate, r , of 0.040, with annual fluctuations in the growth rate, SD (r), of 0.141 (Mirande *et al.* 1993). Life-table analysis predicted an r of 0.052. Simulations using VORTEX predicted an r of 0.046 into the future, with a SD (r) of 0.081. The lower growth rate projected by the stochastic model reflects the effects of inbreeding and perhaps imbalanced sex ratios among breeders in the simulation, factors that are not considered in deterministic life-table calculations. Moreover, life-table analyses use mean birth and death rates to calculate a single estimate of the population growth rate. When birth and death rates are fluctuating, it is more appropriate to average the population growth rates calculated separately from birth and death rates for each year. This mean growth rate would be lower than the growth rate estimated from mean life-table values.

When the simulation model was started with the 18 cranes present in 1938, it projected a population size in 1991 ($N \pm SD = 151 \pm 123$) almost exactly the same as that observed ($N = 146$). The large variation in population size across simulations, however, indicates that very different fates (including extinction) were almost equally likely. The model slightly underestimated the annual fluctuations in population growth [model SD (r) = 0.112 v. actual SD (r) = 0.141]. This may reflect a lack of full incorporation of all aspects of stochasticity into the model, or it may simply reflect the sampling error inherent in stochastic phenomena. Because the data input to the model necessarily derive from analysis of past trends, such retrospective analysis should be viewed as a check of consistency, not as proof that the model correctly describes current population dynamics. Providing another confir-

mation of consistency, both deterministic calculations and the simulation model project an over-wintering population of whooping cranes consisting of 12% juveniles (less than 1 year of age), while the observed frequency of juveniles at the wintering grounds in Texas has averaged 13%.

Convincing evidence of the accuracy, precision and usefulness of PVA simulation models would require comparison of model predictions to the distribution of fates of many replicate populations. Such a test probably cannot be conducted on any endangered species, but could and should be examined in experimental non-endangered populations. Once simulation models are determined to be sufficiently descriptive of population processes, they can guide management of threatened and endangered species (see above and Lindenmayer *et al.* 1993). The use of PVA modelling as a tool in an adaptive management framework (Clark *et al.* 1990) can lead to increasingly effective species recovery efforts as better data and better models allow more thorough analyses.

Directions for Future Development of PVA Models

The PVA simulation programs presently available model life histories as a series of discrete (seasonal) events, yet many species breed and die throughout much of the year. Continuous-time models would be more realistic and could be developed by simulating the time between life-history events as a random variable. Whether continuous-time models would significantly improve the precision of population viability estimates is unknown. Even more realistic models might treat some life-history events (e.g., gestation, lactation) as stages of specified duration, rather than as instantaneous events.

Most PVA simulation programs were designed to model long-lived, low fecundity (K-selected) species such as mammals, birds and reptiles. Relatively little work has been devoted to developing models for short-lived, high-fecundity (r-selected) species such as many amphibians and insects. Yet, the viability of populations of r-selected species may be highly affected by stochastic phenomena, and r-selected species may have much greater minimum viable populations than do most K-selected species. Assuring viability of K-selected species in a community may also afford adequate protection for r-selected species, however, because of the often greater habitat-area requirements of large vertebrates. Populations of r-selected species are probably less affected by intrinsic demographic stochasticity because large numbers of progeny will minimise random fluctuations, but they are more affected by environmental variations across space and time. PVA models designed for r-selected species would probably model fecundity as a continuous distribution, rather than as a completely specified discrete distribution of litter or clutch sizes; they might be based on life-history stages rather than time-increment ages; and they would require more detailed and accurate description of environmental fluctuations than might be required for modelling K-selected species.

The range of PVA computer simulation models becoming available is important because the different assumptions of the models provide capabilities for modelling diverse life histories. Because PVA models always simplify the life history of a species, and because the assumptions of no model are likely to match exactly our best understanding of the dynamics of a population of interest, it will often be valuable to conduct PVA modelling with several simulation programs and to compare the results. Moreover, no computer program can be guaranteed to be free of errors. There is a need for researchers to compare results from different PVA models when applied to the same analysis, to determine how the different assumptions affect conclusions and to cross-validate algorithms and computer code.

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Appendix. Sample Output from VORTEX

Explanatory comments are added in italics

VORTEX—simulation of genetic and demographic stochasticity

TEST

Simulation label and output file name

Fri Dec 20 09:21:18 1991

2 population(s) simulated for 100 years, 100 runs

VORTEX first lists the input parameters used in the simulation:

HETEROSIS model of inbreeding depression
with 3.14 lethal equivalents per diploid genome

Migration matrix:

	1	2
1	0.9900	0.0100
2	0.0100	0.9900

*i.e. 1% probability of migration from
Population 1 to 2, and from Population 2 to 1*

First age of reproduction for females: 2 for males: 2

Age of senescence (death): 10

Sex ratio at birth (proportion males): 0.5000

Population 1:

Polygynous mating; 50.00 per cent of adult males in the breeding pool.

Reproduction is assumed to be density independent.

50.00 (EV = 12.50 SD) per cent of adult females produce litters of size 0

25.00 per cent of adult females produce litters of size 1

25.00 per cent of adult females produce litters of size 2

EV is environmental variation

50.00 (EV = 20.41 SD) per cent mortality of females between ages 0 and 1

10.00 (EV = 3.00 SD) per cent mortality of females between ages 1 and 2

10.00 (EV = 3.00 SD) per cent annual mortality of adult females (2 ≤ age ≤ 10)

50.00 (EV = 20.41 SD) per cent mortality of males between ages 0 and 1

10.00 (EV = 3.00 SD) per cent mortality of males between ages 1 and 2

10.00 (EV = 3.00 SD) per cent annual mortality of adult males (2 ≤ age ≤ 10)

EVs have been adjusted to closest values possible for binomial distribution.
 EV in reproduction and mortality will be correlated.

Frequency of type 1 catastrophes: 1.000 per cent
 with 0.500 multiplicative effect on reproduction
 and 0.750 multiplicative effect on survival

Frequency of type 2 catastrophes: 1.000 per cent
 with 0.500 multiplicative effect on reproduction
 and 0.750 multiplicative effect on survival

Initial size of Population 1: (set to reflect stable age distribution)

Age	1	2	3	4	5	6	7	8	9	10	Total
	1	0	1	1	0	1	0	0	1	0	5 Males
	1	0	1	1	0	1	0	0	1	0	5 Females

Carrying capacity = 50 (EV = 0.00 SD)
 with a 10.000 per cent decrease for 5 years.

Animals harvested from population 1, year 1 to year 10 at 2 year intervals:

- 1 females 1 years old
- 1 female adults (2 <= age <= 10)
- 1 males 1 years old
- 1 male adults (2 <= age <= 10)

Animals added to population 1, year 10 through year 50 at 4 year intervals:

- 1 females 1 years old
- 1 females 2 years old
- 1 males 1 years old
- 1 males 2 years old

Input values are summarised above, results follow.

VORTEX now reports life-table calculations of expected population growth rate.

Deterministic population growth rate (based on females, with assumptions of no limitation of mates and no inbreeding depression):

$$r = -0.001 \quad \lambda = 0.999 \quad RO = 0.997$$

Generation time for: females = 5.28 males = 5.28

Note that the deterministic life-table calculations project approximately zero population growth for this population.

Stable age distribution:	Age class	females	males
	0	0.119	0.119
	1	0.059	0.059
	2	0.053	0.053
	3	0.048	0.048
	4	0.043	0.043
	5	0.038	0.038
	6	0.034	0.034
	7	0.031	0.031
	8	0.028	0.028
	9	0.025	0.025
	10	0.022	0.022

Ratio of adult (>=2) males to adult (>=2) females: 1.000

Population 2:

Input parameters for Population 2 were identical to those for Population 1.
 Output would repeat this information from above.

Simulation results follow.

Population 1

Year 10

N[Extinct] = 0, P[E] = 0.000
 N[Surviving] = 100, P[S] = 1.000
 Population size = 4.36 (0.10 SE, 1.01 SD)
 Expected heterozygosity = 0.880 (0.001 SE, 0.012 SD)
 Observed heterozygosity = 1.000 (0.000 SE, 0.000 SD)
 Number of extant alleles = 8.57 (0.15 SE, 1.50 SD)

Population summaries given, as requested by user, at 10-year intervals.

Year 100

N[Extinct] = .86, P[E] = 0.860
 N[Surviving] = 14, P[S] = 0.140
 Population size = 8.14 (1.27 SE, 4.74 SD)
 Expected heterozygosity = 0.577 (0.035 SE, 0.130 SD)
 Observed heterozygosity = 0.753 (0.071 SE, 0.266 SD)
 Number of extant alleles = 3.14 (0.35 SE, 1.29 SD)

In 100 simulations of 100 years of Population 1:

86 went extinct and 14 survived.

This gives a probability of extinction of 0.8600 (0.0347 SE),
or a probability of success of 0.1400 (0.0347 SE).

99 simulations went extinct at least once.

Median time to first extinction was 5 years.

Of those going extinct,

mean time to first extinction was 7.84 years (1.36 SE, 13.52 SD).

123 recolonisations occurred.

Mean time to recolonisation was 4.22 years (0.23 SE, 2.55 SD).

110 re-extinctions occurred.

Mean time to re-extinction was 54.05 years (2.81 SE, 29.52 SD).

Mean final population for successful cases was 8.14 (1.27 SE, 4.74 SD)

Age 1	Adults	Total	
0.14	3.86	4.00	Males
0.36	3.79	4.14	Females

During years of harvest and/or supplementation

mean growth rate (r) was 0.0889 (0.0121 SE, 0.4352 SD)

Without harvest/supplementation, prior to carrying capacity truncation,

mean growth rate (r) was -0.0267 (0.0026 SE, 0.2130 SD)

Population growth in the simulation ($r = -0.0267$) was depressed relative to the projected growth rate calculated from the life table ($r = -0.001$) because of inbreeding depression and occasional lack of available mates.

Note: 497 of 1000 harvests of males and 530 of 1000 harvests of females could not be completed because of insufficient animals.

Final expected heterozygosity was 0.5768 (0.0349 SE, 0.1305 SD)
 Final observed heterozygosity was 0.7529 (0.0712 SE, 0.2664 SD)
 Final number of alleles was 3.14 (0.35 SE, 1.29 SD)

Population 2

Similar results for Population 2, omitted from this Appendix, would follow.

***** Metapopulation Summary *****

Year 10

N[Extinct] = 0, P[E] = 0.000
 N[Surviving] = 100, P[S] = 1.000
 Population size = 8.65 (0.16 SE, 1.59 SD)
 Expected heterozygosity = 0.939 (0.000 SE, 0.004 SD)
 Observed heterozygosity = 1.000 (0.000 SE, 0.000 SD)
 Number of extant alleles = 16.92 (0.20 SE, 1.96 SD)

Metapopulation summaries are given at 10-year intervals.

Year 100

N[Extinct] = 79, P[E] = 0.790
 N[Surviving] = 21, P[S] = 0.210
 Population size = 10.38 (1.37 SE, 6.28 SD)
 Expected heterozygosity = 0.600 (0.025 SE, 0.115 SD)
 Observed heterozygosity = 0.701 (0.050 SE, 0.229 SD)
 Number of extant alleles = 3.57 (0.30 SE, 1.36 SD)

In 100 simulations of 100 years of Metapopulation:

79 went extinct and 21 survived.

This gives a probability of extinction of 0.7900 (0.0407 SE),
 or a probability of success of 0.2100 (0.0407 SE).

97 simulations went extinct at least once.

Median time to first extinction was 7 years.

Of those going extinct,

mean time to first extinction was 11.40 years (2.05 SE, 20.23 SD).

91 recolonisations occurred.

Mean time to recolonisation was 3.75 years (0.15 SE, 1.45 SD).

73 re-extinctions occurred.

Mean time to re-extinction was 76.15 years (1.06 SE, 9.05 SD).

Mean final population for successful cases was 10.38 (1.37 SE, 6.28 SD)

Age 1	Adults	Total	
0.48	4.71	5.19	Males
0.48	4.71	5.19	Females

During years of harvest and/or supplementation

mean growth rate (r) was 0.0545 (0.0128 SE, 0.4711 SD)

Without harvest/supplementation, prior to carrying capacity truncation,

mean growth rate (r) was -0.0314 (0.0021 SE, 0.1743 SD)

Final expected heterozygosity was 0.5997 (0.0251 SE, 0.1151 SD)

Final observed heterozygosity was 0.7009 (0.0499 SE, 0.2288 SD)

Final number of alleles was 3.57 (0.30 SE, 1.36 SD)

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