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Global Re-introduction Perspectives: 2013

Further case-studies from around the globe

Edited by Pritpal S. Soorae







IUCN/SSC Re-introduction Specialist Group (RSG)

RSG

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IUCN
IUCN, International Union for Conservation of Nature, helps the world find pragmatic solutions to our most pressing environment and development challenges. IUCN's work focuses on valuing and conserving nature, ensuring effective and equitable governance of its use, and deploying nature-based solutions to global challenges in climate, food and development. IUCN supports scientific research, manages field projects all over the world, and brings governments, NGOs, the UN and companies together to develop policy, laws and best practice. IUCN is the world's oldest and largest global environmental organization, with more than 1,200 government and NGO Members and almost 11,000 volunteer experts in some 160 countries. IUCN's work is supported by over 1,000 staff in 45 offices and hundreds of partners in public, NGO and private sectors around the world.

IUCN Species Survival Commission (SSC)
The SSC is a science-based network of close to 8,000 volunteer experts from almost every country of the world, all working together towards achieving the vision of, "A world that values and conserves present levels of biodiversity."

Environment Agency - ABU DHABI (EAD)
The EAD was established in 1996 to preserve Abu Dhabi's natural heritage, protect our future, and raise awareness about environmental issues. EAD is Abu Dhabi's environmental regulator and advises the government on environmental policy. It works to create sustainable communities, and protect and conserve wildlife and natural resources. EAD also works to ensure integrated and sustainable water resources management, and to ensure clean air and minimize climate change and its impacts.

Denver Zoological Foundation (DZF)
The DZF is a non-profit organization whose mission is to "secure a better world for animals through human understanding". DZF oversees Denver Zoo and conducts conservation education and biological conservation programs at the zoo, in the greater Denver area, and worldwide. Over 3,800 animals representing more than 650 species call Denver Zoo home. A member of the World Association of Zoos and Aquariums (WAZA), Denver Zoo's accreditation from the Association of Zoos and Aquariums (AZA) assures the highest standards of animal care. A leader in environmental action, Denver Zoo was the first U.S. zoo to receive ISO 14001 sustainability certification for its entire facility and operations and in 2011 was voted the greenest zoo in the country. The ISO 14001 international certification ensures the zoo attains the highest environmental standards. Since 1996, Denver Zoo has participated in over 590 conservation projects in 62 countries. In 2011 alone, Denver Zoo participated in 102 projects in 18 countries and spent well over $1.5 million to support of wildlife conservation in the field.

Re-introduction Specialist Group (RSG)
The RSG is a network of specialists whose aim is to combat the ongoing and massive loss of biodiversity by using re-introductions as a responsible tool for the management and restoration of biodiversity. It does this by actively developing and promoting sound inter-disciplinary scientific information, policy, and practice to establish viable wild populations in their natural habitats.

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Shaikha Al Dhaheri,
Executive Director,
Terrestrial & Marine Biodiversity Sector,
Environment Agency - ABU DHABI

It gives us great pleasure and honor in supporting the production of the 4th issue of the *Global Re-introduction Perspectives 2013*. It is exciting to know that those 236 case studies will be read by more than 300 members, practitioners and decision makers throughout the RSG network and beyond, who will get the advantage to use it as a tool and reference for future programs and projects that will combat the continuous loss of species through re-introductions and translocation.

Small or big, success or failure, all these case-studies have surely made a difference in regard to the targeted species. This has been achieved by various means such as stabilizing populations, or re-establishing them, increasing their numbers in *ex situ* collections as they have suffered significant declines or even extinction in the wild.

Species re-introductions are an important feature of global conservation efforts and for the newly developed *IUCN Guidelines for Re-introduction and Other Conservation Translocations* along with this RSG edition will act as a powerful reference worldwide and especially to us in the Environment Agency embarking into new initiatives of re-introduction and translocations.

Finally, I would like to thank all practitioners and conservationists who shared their case-studies with us in this edition for their commitment, dedication and passion towards conserving species. Also I thank Denver Zoological Foundation for supporting RSG efforts, the editor of this edition Mr. Pritpal Soorae, Dr. Frédéric Launay, RSG Chair and Dr. Simon Stuart Chair IUCN Survival Commission for their continued devotion and contribution to species conservation worldwide.

Richard P. Reading,
Vice President for Conservation,
Denver Zoological Foundation

I am honored to have the opportunity to provide a forward to *Global Re-introduction Perspectives: 2013: Further case studies from around the globe* published by the IUCN Re-introduction Specialist Groups (RSG) and edited by Pritpal Soorae. Within this four volume set, Pritpal has pulled together an amazing 236 case studies on a wide variety of taxa from plant to invertebrates to vertebrates from all over the world.

Through these case studies and the recently released *Guidelines for Re-introductions and Other Conservation Translocations* by the Re-introduction and Invasive Species Specialist Groups' Task Force for Moving Plants and Animals for Conservation Purposes, the RSG has produced a valuable set of references for current and future translocation practitioners as they strive to restore populations of species depleted by the growing human footprint on our planet and finite resources.

The Denver Zoological Foundation is proud to support this publication and other RSG efforts to improve re-introduction success throughout the globe. We congratulate Pritpal Soorae on this fine accomplishment and extend our thanks to Dr. Frédéric Launay and the RSG for supporting this important publication, Dr. Simon Stuart and the IUCN Species Survival Commission, The Environment Agency – ABU DHABI, and especially to the contributors to this volume for their excellent summaries of re-introduction case studies from around the world.

Simon Stuart,
Chair,
IUCN Species Survival Commission

It seems like yesterday that I wrote the foreword for the third edition of *Global Reintroduction Perspectives*. Such is the pace of re-introduction efforts that another volume with 52 case studies is now available to inform and guide reintroduction practitioners worldwide. We now have an impressive 236 case studies from the four volumes of *Global Re-introduction Perspectives* published so far. In my previous foreword I recommended setting up a searchable database on the RSG website comprising all the case studies. I understand that steps are now being taken to implement this suggestion, and this will, I am sure, make the information in this excellent series much more broadly available to support the work of practitioners.

As in previous volumes, there is impressive taxonomic and geographic coverage in this latest edition. This ability to collect information on re-introductions worldwide is only possible because of the long-term focus and activity of the Re-introduction Specialist Group (RSG) of the IUCN Species Survival Commission. While this fourth edition was being prepared, the RSG completed the new *IUCN Guidelines for Re-introductions and Other Conservation Translocations*, which will provide further impetus to the efforts to return species to parts of their native ranges from which they had been lost.

As with the previous issue, I thank: the Environment Agency Abu Dhabi (EAD), in particular its Secretary General H.E. Razan Khalifa Al Mubarak, for the EAD's long-term and most generous support of the RSG; the Denver Zoological Foundation, in particular Dr Richard Reading, for supporting this publication; the RSG Chair, Dr Frédéric Launay; and the RSG's Programme Officer and editor of *Global Re-introduction Perspectives*, Mr Pritpal Singh Soorae. Without these people, *Global Re-introduction Perspectives* would not be possible.

Frédéric Launay,
Chair,
IUCN/SSC Re-introduction Specialist Group

The IUCN/SSC Re-introduction Specialist Group is glad to present the 4th Issue of Global Re-introduction Perspectives 2013. The series is receiving very good feedback and is gathering momentum under the capable hands of Pritpal Soorae.

A total of 236 case studies of various type of re-introduction, successful or not, have been collected and summarized in the four publications showing the relevance of re-introduction to species conservation. Actually the number of re-introduction projects, feasibility or research/trials is increasing in all taxa.

Whilst it is encouraging to see that re-introductions and translocations are widely used as a conservation tools for many taxa, it is also an indication that the pressure on species is increasing and that quality habitats and space available for species for is decreasing either through direct competition from alternatives land-use or through climate change and its associated effects.

The newly released 2013 *IUCN Guidelines for Re-introduction and Other Conservation Translocations* are addressing this increased reliance and/or application of translocations for species conservation and include reflection and guidance on controversial and debated issues as assisted colonization and ecological replacement. These guidelines are a much needed addition for practitioners and are very fitted for many of the case studies mentioned in this 4th Edition.

The new Guidelines and the cases studies highlighted in that publication are, we hope, a welcome contribution from the Re-introduction Specialist Group to the species conservation array of knowledge tools and prove useful to the practitioners, policy-makers and decision-makers.

I would like to conclude by thanking all the people that contributed case studies, not only for their contributions, but more importantly for their dedication and efforts in working on conserving species worldwide.

An overview and analysis of the re-introduction project case studies

Pritpal S. Soorae, Editor

Introduction

This is the fourth issue in the *Global Re-introduction Perspectives* series and has been produced in the same standardized format as the previous three to maintain the style and quality. The case-studies are arranged in the following order: Introduction, Goals, Success Indicators, Project Summary, Major Difficulties Faced, Major Lessons Learned, Success of Project with reasons for success or failure. For the first issue I managed to collect 62 case-studies, the second issue 72 case-studies, the third issue 50 case-studies and this one 52 case-studies.

These case studies in this issue cover the following taxa as follows:

- Invertebrates - 2
- Fish - 4
- Amphibians - 1
- Reptiles - 3
- Birds - 10
- Mammals - 24
- Plants - 8

I would also like to take this opportunity to thank the various authors for their patience and willingness to submit information on their projects and in many cases with a tight deadline. A few promised articles were not submitted by the last deadline and hopefully if we do another issue we can present them there. We hope the information presented in this book will provide a broad global perspective on challenges facing re-introduction projects trying to restore biodiversity.

IUCN Statutory Regions

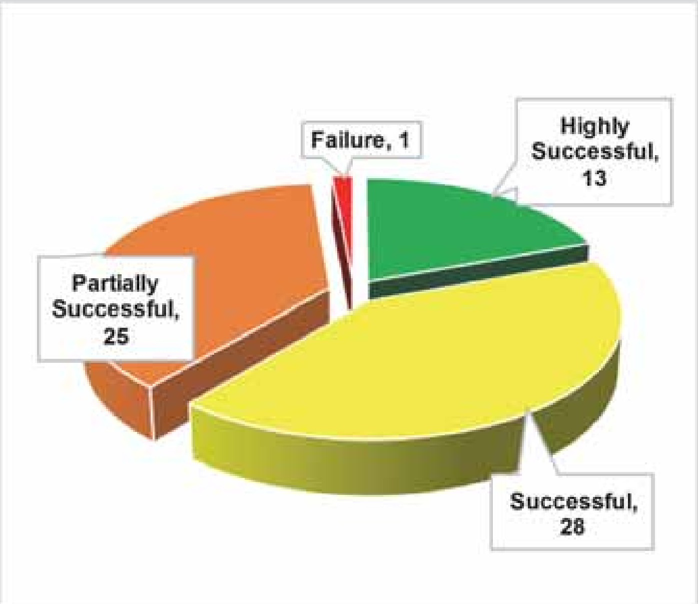
The IUCN statues have established a total of 8 global regions for the purposes of its representation in council. The IUCN's "statutory regions" are a list of States by Region, as per article 16 and 17 of the Statutes and Regulation 36 of the Regulations.

All eight global regions are represented within these case studies and the regions are as follows:
1. North America & Caribbean - 18
2. West Europe - 6
3. South & East Asia - 7
4. Oceania - 6
5. West Asia - 2
6. Africa - 4
7. Meso & South America - 3
8. East Europe, North & Central Asia - 6

Success/Failure of Projects

The projects presented here were ranked as Highly Successful, Successful, Partially Successful and Failure. Out of the 52 case-studies there were a total of 67 releases. In some cases there were multiple rankings as releases were conducted at more then one site or country. In some cases multiple species were released in more than one country. This made analysis difficult but in total the rankings can be seen in figure 1, 13 projects were Highly Successful, 28 were Successful, 25 were Partially Successful and 1 was a Failure.

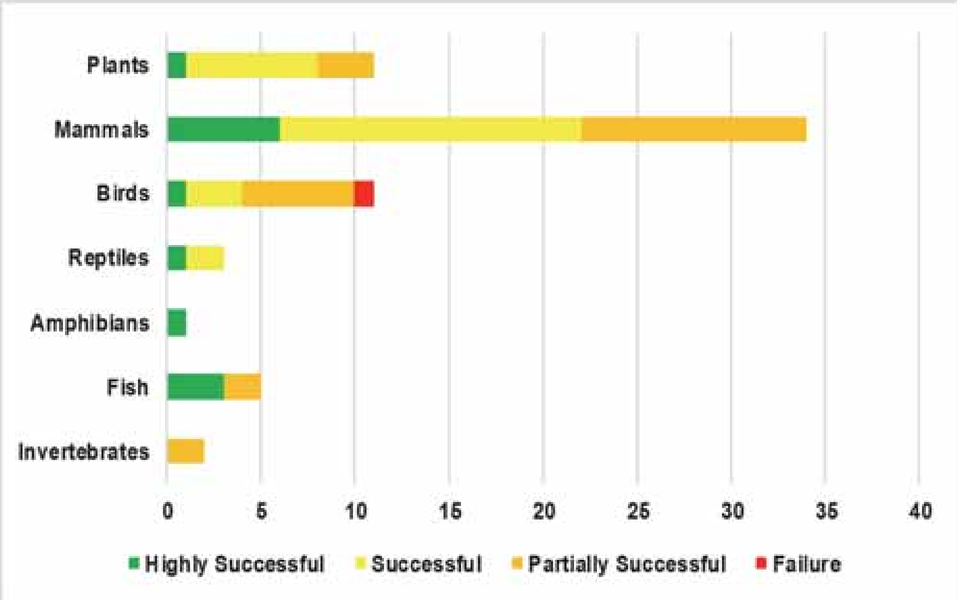
Fig. 1. Success/Failure of re-introduction projects



Success according to the taxa

An analysis was done to gauge the three different levels of success (highly successful, successful, partially successful) and failure

Fig. 2. Success/Failure of re-introduction projects according to major taxa



against the seven major taxa i.e. invertebrates, fish, amphibians, reptiles, birds, mammals and plants as can be seen in figure 2. Out of the seven major taxa only invertebrates did not have a project ranked as highly successful. There was only one amphibian case study and this was ranked as highly successful. The bird projects had all four rankings. The majority of plant and mammal projects were successful and the birds had a majority of partially successful projects.

Future issues of *Global Re-introduction Perspectives*

If you need any further information on future issues issue please contact me for further details. We would also appreciate any feedback you may have from this book. The Editor can be contacted at: iucnrsg@gmail.com

Translocation and augmentation of the fen raft spider populations in the UK

Helen Smith[1], Dave Clarke[2], David Heaver[3], Ian Hughes[4], Paul Pearce-Kelly[5] & Tony Sainsbury[6]

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Introduction

The fen raft spider (*Dolomedes plantarius*) is one of Britain's largest, most spectacular and rarest spiders. Thought to be confined to just three lowland wetland sites (two in England, one in Wales) its UK distribution is highly disjunct. Because of degradation and loss of wetland habitat over much of its range it is the only European spider species to be listed as internationally Vulnerable (IUCN, 1996), as well as being Red Listed by many European countries. In the UK it is regionally listed as Endangered (Bratton, 1991) and is one of only two spider species fully protected by law (Wildlife and the Countryside Act 1981) and is listed under Section 41, NERC Act 2006. It has been the subject of a Natural England Species Recovery Program (SRP) since 1991, and a Species Action Plan since 1999. One of the SRP targets in England is to establish additional populations by translocation. This was considered necessary because of the small number of unconnected populations (one of which is extremely vulnerable: Smith, 2000 & 2013), a propensity for low dispersal (Pearson, 2008), and fragmented distribution of suitable habitat. This target augments and influences habitat management work to secure the existing populations.

Adult fen raft spider in natural habitat © Helen Smith

Goals

- Goal 1: The overall goal is to secure the future of *D. plantarius*

Fen raft spider release in 2012 © Ian Hughes

as a UK species by increasing the number of sustainable populations from the current three to 12 by 2020. This to be realized by a combination of direct translocations and captive rearing initiatives.

- Goal 2: Identify the most appropriate provenance of UK stock for release at new sites over two successive years to create populations with a natural age structure.
- Goal 3: Identify new receptor sites, with appropriate management of suitable habitat already in place, that also have high levels of landscape-scale connectivity to allow natural expansion of new populations.
- Goal 4: Consider impacts of climate change in the selection of new sites. The previous UK range of this species is not known (it was not described in the UK until 1956). New host sites are being considered not only within the range described by existing populations but also further north; the UK is mid-latitudinal range for this species.
- Goal 5: Subsequently monitor population size, range increase and genetic diversity. This involves convening a steering group of stakeholders and experts to advise on and oversee this process.

Success Indicators

- Indicator 1: Pre-program work confirms understanding of this species current status.
- Indicator 2: Identification of sufficient host sites with both appropriate connected habitat and guaranteed continuity of sympathetic management and ownership, in appropriate geographical areas.
- Indicator 3: Collaboration between all relevant government agencies, NGOs, landowners and other stakeholders including establishment of a multi-disciplinary steering group.
- Indicator 4: Assurance of ability of largest surviving UK population to sustain annual harvesting of 10 - 20 adult females for direct release and of smallest population to sustain removal of 5 - 10 adult females for the *ex situ* rearing program with associated protocols for *ex situ* biosecurity and pre- and post-release disease monitoring.
- Indicator 5: Appropriate post-release monitoring program in place to ensure long term target of self-sustaining populations expanding in range on 12 UK sites.

Project Summary

Feasibility: Field surveys by British Arachnological Society volunteers, initiated three years before the recovery began, confirmed the current status of *D. plantarius* and identified potential new English receptor sites. Confirmation of species' status also came from checking records of its congener *D. fimbriatus*, with which it is easily confused, and from a public appeal for *Dolomedes* sightings. Field survey work is ongoing.

Implementation phase: By 2010 receptor sites had been identified and prioritized, and work undertaken to establish appropriate provenance of stock. The priority sites were on the same river system as the most vulnerable extant population but with habitat much more similar to that of the more distant English population. Although these two populations differed genetically, no evidence was found of either inbreeding depression or hybrid vigor when they were crossed in captivity (Smith, 2011). It was therefore decided to stock new sites from both populations to maximize genetic diversity. Introductions to new sites of spiders from the smaller and more vulnerable of the English populations (Smith, 2000) used three-month-old spiderlings that had been captive-reared in individual test tubes. With mean brood size of over 500 and survival to three months in captivity of >80%, large numbers of spiderlings were available for release from relatively small numbers of wild-caught females. These females were also retained in captivity until they produced second broods (this species stores sperm) before re-release at their point of capture. Successful second brood production is significantly greater in captivity than in the wild and so this method helped to offset depletion of the source population. Females from the larger English population were also caught from the wild carrying egg sacs and were retained in captivity until their broods hatched. They were then released to new sites with their broods (this species shows maternal care), removing the risk of predation of the female and her egg sac.

As well as establishing new site populations, the translocation program sought to expedite recolonization of recently restored habitat within the more vulnerable English site. The population there had become confined to two small areas and had undergone a sharp decline in genetic diversity over a 20 year period (Holmes, 2008). Captive reared stock from these residual areas was released into two areas of restored habitat between 2010 and 2012 and a chain of new ponds excavated to help

Spider release into the wild © Sheila Tilmouth restore hydrological connectivity between them. A conservative approach was adopted, using stock only of local provenance for these re-introductions.

Much of the captive rearing of spiderlings was undertaken by a consortium of UK zoos and collections recruited through the British and Irish Association of Zoos and Aquaria Terrestrial Invertebrate Working Group, operating under a biosecurity protocol developed by the Zoological Society of London (ZSL) (Hopkins & Sainsbury, 2013). An ongoing pathological study by ZSL of unusual mortality events during captive-rearing is informing further development of the rearing protocol. All of the work on this species is subject to license in the UK under the Wildlife and the Countryside Act 1981 and all aspects of sourcing and releasing the spiders on sites designated as Sites of Special Scientific Interest are conditional on formal Natural England consents under this Act and the Countryside and Rights of Way Act 2000.

Post release monitoring: By 2012, introductions had been made in two successive years at two sites on the same river system and a first introduction had been made at one more distant site. Successful breeding was confirmed at the first two sites, from spiders released in 2010, with nursery web densities similar to the highest encountered in the source populations. Because of this, there are no current plans for augmentation at these sites. The process of site assessment and translocations is ongoing with monitoring results informing development of the methodology. If the new populations continue to thrive, captive rearing from the fragile population is likely to be a short-lived phase. Subject to the results of genetic monitoring, future stock for translocation is likely to be harvested from the new populations. Successful breeding has also been confirmed in the releases at the most vulnerable natural site but quantitative assessment of success is more problematic because the habitat is much more difficult to monitor. This species always attracts a high level of UK media attention and the breeding and release program gained extensive national and international media coverage. The involvement of 10 UK zoos in 2012 also brought the work to the attention of a large new audience. The public profile of the project is being used to promote awareness not only of the plight of this species but also of the many other rare species suffering from the loss of lowland wetland habitats. In addition, the work has provided a focal point for those working on the conservation of this species throughout its European range. It continues to increase understanding of the species' biology that unpins effective conservation delivery.

Major difficulties faced

- The captive rearing work was initially undertaken only by the project coordinator but the increasing participation of UK zoos and collections made this labor-intensive aspect of the program much more viable, with groups of 100 – 200 spiderlings being taken per institution.
- Problems with generating an appropriate food supply for the captive-reared spiders without resort to buying-in from the live-food industry with accompanying pathogen/parasite risks.

- Problems with annual funding uncertainties for the longer-term program, particularly for the critical post-translocation monitoring phase.
- Lack of funding for monitoring necessitates dependence on volunteers. The difficulties both of recruiting sufficient volunteers, and of designing robust protocols that can be delivered reliably, threaten effective delivery of the original monitoring goals.

Major lessons learned

- Ease of post-translocation monitoring, and consequent likelihood of detection of successful establishment, is highly dependent on habitat accessibility. Because of this the program is now concentrating on habitat where success is easily proven before moving on to habitat in which success is also likely but where detection is more difficult.
- Although spiders are often recipients of negative press coverage, the huge media and public response to this translocation program was overwhelmingly positive and an excellent platform for promoting wider issues around the conservation of spiders and other invertebrates.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- The *D. plantarius* translocation program is at a relatively early stage and ongoing. Preliminary indications are that the introductions are highly successful on some sites but more difficult to assess on others.
- Any success is due to adherence of the program plan, protocols and associated funding support.

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The translocation of the red barbed ant from the Isles of Scilly to Chobham Common National Nature Reserve, Surrey, UK

Scotty Dodd[1], Nicola Gammons[2], Dave Clarke[3] & Paul Pearce-Kelly[4]

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Introduction

The red barbed ant (*Formica rufibarbis*) RDB1 UKBAP/SPI (S41) is possibly the rarest resident animal in mainland Britain (Pontin, 2005). Known from two lowland heath nest sites at Chobham Common in Surrey on the mainland, with a strong but much localized population extant at St. Martins, Isles of Scilly. Extensive restoration work was completed at the release site prior to the ants return. This *Heritage Lottery Fund* supported project enabled Chobham Common site management and collection of freshly mated *F. rufibarbis* queens from the St. Martins population for translocation to their National Nature Reserve Chobham Common release site in Surrey. The translocation plan necessitated an interim *ex situ* rearing element to enable the young queens to produce initial attendant workers to improve colony establishment chances. Although the project successfully realized it's technical remit components through to field release, greater than envisaged competitor ant pressure, especially *Lasius niger*, compromised the fledgling released *F. rufibarbis* colonies. This demonstrated a need for release larger colonies to enable release colonies to repel the competitor species. This requirement likely necessitates whole nest translocations. Genetic analysis conducted as part of the project confirmed the closest related European populations for future translocation initiatives.

Red barbed ant © Richard Alan

Goals

- Goal 1: Establish self-sustainable populations within historic range of Surrey.
- Goal 2: Realize a successful captive maintenance regime for temporarily maintaining *ex-situ* ant colonies prior to release, along with health screening protocols.

- Goal 3: Conduct long-term monitoring of extant and introduced populations, including targeted surveys to identify new, previously undetected colonies.
- Goal 4: Continue research into autecology and genetics.

Red barbed ant colony at ZSL

Success Indicators

- Indicator 1: Long-term establishment of introduced nests at target sites.
- Indicator 2: Introduced nests producing sexuals, which successfully mate and go on to produce new nests.
- Indicator 3: Successful healthy captive maintenance between initial colony collection and release into selected sites.
- Indicator 4: Publish survey and monitoring results as annual online reports.
- Indicator 5: Publication of research reports, academic dissertations and peer-reviewed papers.

Project Summary

Feasibility: A fairly common species throughout Europe. Apparently declined to only two nests at a single heathland site, Chobham Common, on mainland Britain. In Britain there is a strong association with heathland habitat, though found to be more catholic in rest of range. Open, early successional habitat with dry, light soils and bare patches appears to be essential.

Implementation: Mated dealate queens harvested from Isles of Scilly over successive summers and taken to isolated quarantine facilities at Zoological Society of London (ZSL) London Zoo for health checks, maintenance and rearing of colonies for release. Rearing of this species had not been achieved in a zoo before, so protocols had to be developed from previous amateur rearing efforts. A full quarantine protocol was implemented to minimize disease and screen for pathogens pre-release. Early successional habitat creation implemented at Chobham Common to act as receptor sites for new colonies. Staggered release of small colonies over several summers.

Post-release monitoring: As far as limited annual field visits for monitoring of released colonies, extant nests and up until 2011 disease risk assessment checks with ZSL vets. Monitoring consists of visual searches for foraging workers at release sites and checking under nest tiles, recording all ant species present. After initial positive results there has been no target species activity recorded at

Releasing *ex situ* reared red barbed ant colonies near Surrey in 2008 © Paul Pearce-Kelly

release sites for over two years, suggesting the releases may have failed or that nests have moved out of the survey areas. Monitoring also incorporates mapping and surveillance of slave-maker ant (*Formica sanguinea*) nests to ensure that the 100m buffer zone around release sites and extant nests is not breached. Results suggest that although still a potential threat slave-makers pose a lower risk than the common black ant (*Lasius niger*), which is abundant at the site and quick to colonise bare ground patches created for the target species. Slave-makers appear to be less abundant at Chobham Common than on other nearby heathland sites.

Major difficulties faced

- Rearing sufficiently large *F. rufibarbis* colonies from mated Queens in captivity over short time (i.e. single season) periods.
- Establishing nests in wild due to heavy completion from *Lasius niger* and, to a lesser degree, *F. sanguinea.*
- Post-release monitoring funding in the long-term (i.e. post HLF funding).

Major lessons learned

- Queen *F. rufibarbis* ants and young workers can be successfully kept in *ex situ* conditions. However, population growth was minimal due to the constraints of same season release schedules.
- *Formica sanguinea* was believed to be the biggest threat to fledgling released colonies - however, the common black ant (*Lasius niger*) proved to be the greatest threat. Because of the difficulties associated with rearing the larger *F. rufibarbis* colonies necessary to repel aggressive ant species it is felt that whole nest translocations are a better option for the future.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- Small colony size released.
- Unexpected competition/aggression from *Lasius niger.*

- Limited resources for post-release monitoring.

Overview of release site © Paul Pearce-Kelly

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Examining genetic diversity, outbreeding depression, and local adaptation with slimy sculpin re-introductions in the southeast Minnesota Driftless Region, USA

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Introduction

The slimy sculpin (*Cottus cognatus*) is a small fish that occupies benthic, cold-water habitats in small streams of the Driftless Region in Minnesota, USA. They are often locally abundant where they are present and are considered an important component of the ecosystems in which they are native. Populations of slimy sculpin were present in this region historically, but many were extirpated as a result of poor land use practices. This re-introduction project aimed to re-establish the slimy sculpin to a portion of its former range in the Driftless Region, but the re-introduction environments, although they were improved, have been substantially modified by humans. A challenge was to decide whether the existing genetically distinct source populations should be matched to a set of local conditions at the re-introduction sites or be mixed to provide greater genetically-based adaptive potential in anthropogenically affected (disturbed) environments (Huff *et al.,* 2010).

Close-up of a slimy sculpin
© David Huff / Lorissa Fujishin

Mixed-source re-introductions are thought to be advantageous in disturbed environments, but they have drawbacks because unique evolutionary lineages should be preserved as much as possible to preserve genetic diversity. Our research investigated the persistence and fitness-related traits of multi-source re-introduced populations of slimy sculpin in the Driftless Region (Huff *et al.,* 2011).

Goals

- Goal 1: Re-establish the slimy sculpin to nine isolated locations within its former range in the Driftless Region of southeast Minnesota and ensure population viability, long-term persistence in the face of environmental change and preserve the evolutionary processes that sustain genetic diversity.

- Goal 2: Characterize patterns of success or failure in re-introduced populations to identify conditions that lead to successful population establishment.
- Goal 3: Evaluate allelic richness and heterozygosity in the re-introduced populations relative to computer simulated expectations.
- Goal 4: Examine how fitness surrogates such as body size, growth rate and body condition differ by ancestral origin in the re-introduced populations and investigate the consequences of outbreeding in first- and second-generation inter-source hybrids.

Success Indicators

- Indicator 1: Establishment of slimy sculpins at all sites for at least three generations and substantial expansion of each population's range away from the re-introduction location.
- Indicator 2: Characterize habitat at each re-introduction site and compare population size estimates with different habitat features such as stream temperature, substrate type, etc.
- Indicator 3: Evaluate the allelic richness and heterozygosity in the re-introduced populations relative to computer simulated expectations.
- Indicator 4: Document how fitness surrogates such as body size, growth rate and body condition differ by ancestral origin in the re-introduced populations and investigate the consequences of outbreeding in first- and second-generation inter-source hybrids.

Project Summary

Feasibility: Brynildson and Brynildson (1978) demonstrated the feasibility of sculpin re-introductions by documenting the establishment and dispersal of sculpins in a Southwest Wisconsin stream. Following a one-time stocking of 500 individuals, stocked sculpins gradually expanded throughout the suitable areas of the stream over the course of eight years. In recent years, the Minnesota Department of Natural Resources (MNDNR) and other organizations completed many stream habitat improvement projects that stabilized eroding banks, improved substrate, increased fish cover, and increased riparian tree abundance to provide shade in the summer. The recipient streams were chosen from among these restored sites because they comprised suitable habitat (coarse-substrate, plentiful riffles and groundwater input) and were repeatedly sampled and verified not to have any sculpin species present. Most of the recipient sites were located on private land; therefore, we contacted landowners and coordinated access for repeated research and monitoring site visits. All of the landowners were amicable and some were interested in the research and also wished to accompany researchers during site visits. Several cold-water streams with abundant sculpin populations were identified as potential donor sources. These locations were surveyed for at least three consecutive years to verify that source populations would not be detrimentally affected by removal of sculpins for stocking. Disease testing was necessary to verify that sculpins to be translocated would not transmit any pathogens to organisms in recipient streams. The MNDNR required three


Slimy sculpin showing coded tag © David Huff

years of negative tests from the donor streams before sculpins could be translocated.

Implementation: To avoid disrupting spawning or stressful handling and transport during hot weather, we collected sculpins for translocation in late October. Sculpins were collected from an established set of locations at each of the three donor streams by backpack electrofishing. The entire designated donor stream reach was sampled at each collection event to provide data for population assessments. Each sculpin was weighed, measured, and marked by clipping a pelvic fin so that stocked fish could be distinguished from naturally reproduced fish. We also tagged several hundred of the sculpins in each of the recipient sites with unique identifiers to collect information about growth, survival, and movement of individual fish. Approximately 150 fish with roughly equal proportions from source streams were translocated in each year from 2003 - 2005 to nine different recipient sites. Specific quantities and timing of translocation activities may be found in Huff *et al.* (2010).

Post-release monitoring: We monitored the establishment of sculpins at each of the recipient sites by sampling sculpins at least once per year in the autumn through 2009. We collected data for population estimates and we tracked the expansion of sculpin presence away from the original re-introduction location. Sculpins spawn once per year in the spring, so we monitored these re-introduced populations for at least four generations. We documented established sculpin populations at all re-introduction sites (Huff, 2010). In 2009 population estimates ranged from 200 to 3,100 sculpins across all nine sites. In some cases the range of the re-introduced sculpins expanded to the extent of the local drainage basin and in other cases sculpin presence remained highly localized near the original stocking site.

We completed habitat surveys at the source and recipient sites in which we characterized substrate type, aquatic macroinvertebrate (an important sculpin prey) abundance and composition, water velocity, water temperature (using data-recording temperature probes), and other habitat features. Based on our results we hypothesized that thermal regime differences between the source habitats provided potential mechanisms for local adaptation development among source populations. Dissimilar optimal growth temperature ranges or maximum growth rate differences may have arisen between source populations as a compensatory

Re-introduction site showing restored riparian corridor © David Huff

response to different temperatures and growing season lengths (Huff, 2010 & Huff *et al.,* 2011). We evaluated allelic richness and heterozygosity in the re-introduced populations relative to computer simulated expectations. Sculpins in re-introduced populations exhibited higher levels of heterozygosity and allelic richness than any single source, but only slightly higher than the single most genetically diverse source population (Huff *et al.,* 2010).

We inferred the relative fitness of different pure strain and hybrid-cross descendants in the re-introduced populations by comparing their growth rate, length, weight, body condition and persistence in re-introduced populations. Pure strain descendants from a single source population persisted in a greater proportion than expected in the re-introduced populations. Length, weight and growth rate were lower for second-generation intra-population hybrid descendants than for pure strain and first-generation hybrids (Huff *et al.,* 2011).

Major difficulties faced

- This project was difficult to fund because the purpose of the re-introduction was to restore ecological integrity to streams and to potentially provide forage for game fish (trout), rather than to establish a threatened species.
- It was often problematic to recruit and organize volunteers to complete the majority of the field work.
- Because the slimy sculpin was a poorly understood species, we found it initially challenging obtain consistent and reliable information regarding its life-history and species identification.
- For genetic analyses, new microsatellite markers for slimy sculpins had to be developed for the first time by our research team (Fujishin *et al.,* 2009).
- It was difficult to identify enough re-introduction sites to meet our research needs.

Major lessons learned

- If feasible, the genetic and ecological distinctiveness of candidate source populations should be evaluated prior to re-introduction.
- Computer simulations may allow the genetic diversity benefits of mixing populations to be weighed against the risks of outbreeding depression in re-introduced and nearby populations.
- Given the absence of information regarding deep phylogenetic separation among populations, the high degree of genetic differentiation, the potential for

disrupting beneficial adaptations, and the lack of evidence that genetic rescue is necessary, the most conservative option available for future re-introductions of slimy sculpin in the Driftless Region would be to use a single source population.
- Single-source re-introductions may be carried out on a trial basis using local strains that maximize the likelihood of genetic and ecological similarity to inhabitants of the surrounding area.
- Monitoring the populations may identify the need to supplement the re-introduced populations with individuals from different strains if it is warranted.

Success of project

Highly Successful	Successful	Partially Successful	Failure
√			

Reason(s) for success/failure:
- We devised thorough plans for re-introducing the sculpins and kept detailed records regarding how many fish were re-introduced, where they came from originally, when and where they were translocated, and we designed a comprehensive population monitoring plan.
- We were careful to translocate fish when the weather was cool and we took other measures to minimize stress on the sculpins.
- We carefully selected re-introduction sites that utilized data from detailed habitat surveys.
- We solicited information and suggestions from local fisheries professionals and academics that significantly improved our project.

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Captive-breeding, re-introduction and supplementation of the European mudminnow in Hungary

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Introduction

The European mudminnow (*Umbra krameri*, Walbaum 1792*:* Actinopterygii: Esociformes: Umbridae) is a relict and endemic species living mainly in marshes and fens in the catchment area of the Danube and Dniester (Tatár *et al.*, 2012). The species is categorized as Vulnerable on the IUCN Red List (IUCN, 2012) and is listed on the Annex II of the European Union Council Directive on the Conservation of natural habitats and of wild fauna and flora, the Appendix II of the Bern Convention and the Red List of many European countries including Hungary. In the latter country the species is listed as strictly protected.

European mudminnow
© Csaba Posztós/Photomania

The European Mudminnow Conservation Program is taking place mainly near Szada, a little village 25 km from Budapest. Between 2008 and 2012, we created seven isolated little ponds fed by groundwater ("Illés Ponds", GPS: N 47° 37' 37,02"; E 19° 17' 31,83") with irregular shores and bottoms in the area (volumes: 50 - 60 m^3, surfaces: 30 - 40 m^2, mean depths: 1 - 1.5 m, max. depth: 2.5 m.) at locations where the vegetation was degraded (so we did not alter important habitat).

Goals

- Goal 1: To investigate natural habitats and ecological needs of the European mudminnow.
- Goal 2: Creation and monitoring of new habitats (ponds) with regard to the results of Goal 1.
- Goal 3: To create European mudminnow breeding stock in the new habitats with the release of captive bred individuals (*in situ* and *ex situ* conservation).
- Goal 4: Supplementation at historic and recent natural habitats with the release of captive bred individuals.
- Goal 5: Cryopreservation of European mudminnow sperm for later breeding (*ex situ* conservation).

Success Indicators

- Indicator 1: Creation of new habitats (“Illés’ Ponds”) in the Model Area of Szada and complex monitoring of new and natural habitats.
- Indicator 2: Artificial propagation of the European mudminnow and cryopreservation of sperm.
- Indicator 3: Survival of the captive bred and released individuals in new habitats.
- Indicator 4: Breeding of the released individuals in the new habitats.
- Indicator 5: Releases to supplement natural populations with captive bred individuals and with those born in the new habitats.

Project Summary

Feasibility: The species is known to have been extirpated from many of its original habitats. It is estimated that mudminnow populations have declined by more than 30% in the past 10 years. The main reason for this decline is considered to be habitat destruction, especially channelization followed by the destruction of river and stream floodplains (Wanzenböck, 1996). Recently, the invasive and aggressive Amur sleeper (*Perccottus glenii*, Dybowski, 1877) supplants *U. krameri* in Hungarian waters. For instance three original European mudminnow habitats were monitored in Hungary in 2010 and in two of them we could catch only Amur sleepers and no *U. krameri*. Systematic stockings of mudminnows into adjacent streams, canals and still waters might help to develop self-sustaining populations of *U. krameri* in places where the species disappeared or occurs only sparsely. The best method for the maintenance of populations would be the preservation of a variety of suitable micro-habitats. Furthermore, artificial propagation of mudminnow could also help to increase its stocks (Bíró & Paulovits, 1995).

The main objectives of the European Mudminnow Conservation Program are the *in situ* and *ex situ* protection of *Umbra krameri* in order to preserve and increase natural stocks.

Release in the Model Area of Szada © Bálint Bajomi

Long-term goals of the program:

- Reconstruction of wetlands and creation of still waters to improve ecological conditions and increase the number of mudminnow habitats in Hungary and other countries.
- Sperm cryopreservation for gene bank and artificial propagation for stocking to sites in Hungary and other countries.
- Genetic research on different populations in the distribution area of genus *Umbra*.
- Monitoring of water quality, zooplankton, macro-invertebrate, macrophyte and fish populations in natural habitats of mudminnow and Amur sleeper (*P. glenii*).
- Monitoring in reconstructed and new (artificially created) habitats.
- Developing a method for the control of spreading of the invasive Amur sleeper.
- Developing Model Area of Szada: creating further separated ponds for *U. krameri* and other endangered marsh fish populations (e.g. *Misgurnus fossilis, Carassius carassius & Leucaspius delineatus*).

Implementation

Results of the first five years (2008 - 2012):

- Seven new ponds ("Illés" ponds) fed by groundwater were created in the Model Area of Szada (average sizes of water surfaces and mean depths 30 - 40 m^2, 1 - 1.5 m).
- The majority of water quality indicators of three years old ponds has reached the characteristic values experienced in natural habitats of the mudminnow.
- The majority of the quantities and numbers of zooplankton and macro-invertebrate species of Illés' ponds have reached the characteristic values experienced in natural habitats of mudminnow in two years.
- Reproduction in captivity, embryo and larva development of European mudminnow were investigated in the labs of the Department of Aquaculture, Szent István University, Gödöllő. Apart from them, the possibilities of pre-nursring and rearing were investigated in controlled condition and artificial pond.
- Propagation and larvae rearing may help in strengthening population considerably, thus supplementing decreased stocks and ponds of Model Area of Szada.


Captive bred juvenile © Csaba Posztós / Photomania

- Stocking of broodfishes in natural habitats and Model Area of Szada (stocked fishes grew faster in the artificially created ponds than literature sources describe).
- We have three different rescued stocks of *Umbra krameri* in four ponds of Model Area of Szada (these separated ponds serve as refuges of endangered Hungarian mudminnow populations).
- Stocked mudminnows spawned in two years old ponds in spring 2011 and 2012.
- We created a European Mudminnow Database which contains biological and ecological data about mudminnow and its habitats.

Post-release monitoring
Results of the post-release monitoring:
- <u>Indicator 1</u>: Physio-chemical, hydrobiological fish fauna and botanical data in seven new ponds and 10 natural habitats.
- <u>Indicator 2</u>: A total of 42 female mudminnows saved from endangered habitats and 864 reared individuals.
- <u>Indicator 3</u>: The population persisted in all four new water bodies where releases took place.
- <u>Indicator 4</u>: Breeding occurred at three release areas among four.
- <u>Indicator 5</u>: We supplemented populations at three natural habitats with 864 captive-bred individuals and 257 fish coming from the wild-born generation of the 3rd Illés pond. They had altogether a conservation value of about US$ 1,225,181 (In Hungary, individuals of species protected under national law have a conservation value in money fixed by law. This is used e.g. when punishing people killing protected animals. The value of one mudminnow individual is US$ 405).

Major difficulties faced
- Fundraising is a difficult issue, so the long term planning and implementation of the project is uncertain (there are no funding opportunities in the public sector giving bigger amounts for several years in Hungary). We tried to apply for international funds, but we did not succeed because our project was considered as of local importance.
- The long process of applying for permits cause difficulties, because we can run out of time at the end of the breeding period.

- Local inhabitants have released *Cyprinidae spp.* and a predatory European perch *(Perca fluviatilis)* into one of the new ponds at Model Area of Szada. We could not catch the latter fish, so it has damaged the European mudminnow and Crucian carp *(C. carassius)* populations of the pond. Wide information dissemination among local inhabitants is not necessarily a good solution to this problem - if more people know about the ponds, more can harm them.
- Reproduction strategy of *U. krameri* is to rear relatively small amount of larvae (100 - 250/female). Contrary to other fish species the artificial propagation methods (for instance using hormone administration for induction of ovulation) are not effective with this fish species so we had to develop new captive breeding methods.

Major lessons learned

- The Model Area of Szada chosen for creating new habitats was ideal, because it is not under legal protection, so applying for permits was easier. It is in vicinity of a species-rich Natura 2000 protected area, with an existing population of the European mudminnow.
- Creating several little habitats instead of one big increased the success of the project. Despite the fact that some of the ponds are close to each other (within 25 m), they all provide different conditions for life, so we could choose those which had high potential for fish survival. Moreover in little ponds monitoring is more efficient, has lower costs and removal of potentially establishing invasive fish species is cheaper. The natural self-purification potential of the created water bodies was high already after a short period of time: in the ponds with high nitrogen compounds (nitrate, nitrite & ammonium) concentration, the nutrient concentration decreased from 97.4% to 76% in 3 years.
- The European mudminnow has an opportunistic alimentation and wide tolerance to water quality. Its decline is due mainly to the draining of fens and marshes, so its populations can be increased with the creation of new water bodies. In 14 - 22 months after the creation of the ponds, the European mudminnow can be released in security, because a suitable food base becomes available. The year after release the fish can already breed.
- According to our studies and other Hungarian and foreign investigations, the invasive and predatory Amur sleeper is a major danger to the mudminnow populations. To make it more difficult for the establishment of invasive fish species, the ponds at the Model Area of Szada are fed by ground water and they do not have connections with each other and different surface water bodies.
- The advantage of captive breeding is the possibility to raise more healthy juveniles and release them to several habitats. The disadvantage is that in order to preserve genetic diversity it is possible to release many juveniles descending from a few parents only at a young age (adaptation, selection), or only a few individuals at older age to avoid potential inbreeding depression. This area needs further study in the near future.

Success of project

Highly Successful	Successful	Partially Successful	Failure
√			

Reason(s) for success/failure:

- Extensive collaboration among different NGOs (e.g. Tavirózsa and Nimfea Associations - Hungary, Umbra Association - Slovakia), Directorates of National Parks, Universities, authorities and Government Institutes, Local government of Szada village, "VITUKI" Institute (ceased operation from 2012) and media (national and local TVs, radios, gazettes etc.).
- Organization of field and lab work (*ex situ* and *in situ* conservation) in harmony with the life cycle of the European mudminnow.
- According to Seddon (1999), *"we could consider any re-introduction as comprising a sequence of three objectives: the survival of the release generation; breeding by the release generation and their offspring; and persistence of the re-established population, perhaps assessed through extinction probability modelling."* The first elements of this definition is already accomplished: the released generation has survived. The second element partly came true (breeding by the release generation). More time is needed to evaluate further criteria (breeding of the offspring and persistence of the population), so long-term success of the program will be known only a few years later.

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Re-establishment of the natural life histories of Eagle Lake rainbow trout, USA

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Introduction

Eagle Lake rainbow trout (*Oncorhynchus mykiss aquilarum*), is endemic to Eagle Lake and its main tributary Pine Creek of northeastern California, USA. Eagle Lake rainbow trout (ELRT) spend most of their life in Eagle Lake, a large highly alkaline lake. The lake consists of two basins averaging 5 - 6 m deep and the third averaging 10 - 20 m, although lake levels may drop further during drought periods. The shallow basins are uniform in their limnology and water temperatures can exceed 20°C in the summer. The deep basin stratifies, so in late summer most of the trout are in the deeper, cooler water of this basin. Pine Creek is the major tributary to Eagle Lake, is approximately 60 km long, of which about 10 km are perennial. During the summer, upper Pine Creek is a cold spring -fed trout stream, flowing through meadows and open forest, with modest gradients.

ELRT is currently listed as a Species of Special Concern by California Department of Fish and Wildlife (CDFW). The American Fisheries Society considers ELRT to be a threatened species and NatureServe has listed it as “critically imperiled’. The ELRT fits the definition of a threatened species under U.S. Endangered Species Act because it is in danger of extinction throughout its native range as a wild, self-sustaining species.

Goals

Goal 1: Restoration of a wild, naturally-spawning population.
Goal 2: Modification of the weir/dam at the bottom of the main tributary stream to allow free passage.
Goal 3: Improvements to the watershed habitat.

Eagle Lake rainbow trout

Electro-fishing for Eagle Lake rainbow trout

Goal 4: Removal of alien fish in spawning streams.
Goal 5: Reach an agreement with all the agencies and stakeholder groups on the restoration actions.

Success Indicators

- Indicator 1: Re-introduced fish spawn naturally in native streams.
- Indicator 2: Part of the fish run pass through the weir and reach the historically spawning grounds.
- Indicator 3: Water diversions are stopped and habitat restoration actions continue.
- Indicator 4: Alien trout are eradicated from Pine Creek watershed.
- Indicator 5: Agencies and stakeholders group are involved in the project and actively work on the restoration project.

Project Summary

Feasibility: The focus of this project is the restoration of the natural spawning run of ELRT to its historic spawning tributary Pine Creek. ELRT are important to the regional economy where this species lives; the trophy fishery in Eagle Lake attracts anglers, providing significant income to local businesses. The trout are also an important cultural resource for the native Paiute people. However, ELRT have not been able to sustain their populations for more than 60 years because of inaccessible spawning areas, competition from alien trout in perennial spawning and rearing areas, and decreased habitat quality of Pine Creek. The major spawning reaches in Pine Creek have been inaccessible due to habitat degradation and barriers, but restoration actions have considerably improved both upstream habitat and access to spawning areas. However, alien brook trout dominate the headwater spawning and rearing streams and it is unlikely ELRT can persist as natural spawners without eradication of the brook trout. The restoration of the natural life history should bring back wild spawning populations of ERLT, and also a run of considerable cultural importance and greatly benefiting the fishery in Eagle Lake. Likelihood of restoration has been greatly increased in recent years by cooperative efforts of CDFW, US Forest Service, Susanville Indian Rancheria, University of California, Davis, and various stakeholders.

Implementation: For the past 25 years, a Cooperative Resource Management Program has resulted in major improvements to the Pine Creek watershed, greatly reducing the impacts of livestock grazing, eliminating passage barriers, and reducing diversion of water. In 2012, CDFW modified the weir at the mouth of Pine Creek to allow volitional passage of ELRT under high winter/spring flows, while also allowing continued take of spawning fish to support the hatchery program. Movement of fish along 50 km of stream is being monitoring through the use of passive integrated transponder (PIT) tags. Experimental transport of adult fish to spawning areas has demonstrated that successful spawning is possible, especially in Bogard Spring Creek, a tributary to Pine Creek, where the brook trout population has been largely eliminated through annual electrofishing. Successful rearing of juvenile ELRT has been observed in the creek although successful return to the lake has not yet been demonstrated. Some ELRT allowed passage over the weir spawned in the intermittent reaches of Pine Creek and small young of year were observed moving over the weir towards the lake as flows dropped.

Post-release monitoring: The project has shown that fish transported to the upper watershed have spawned and reared successfully. The study has also demonstrated that despite 60 years in captivity, ELRT are still capable of migrating upstream to spawn and of rearing in Pine Creek. These results also indicate that trapping and trucking is a viable option for helping to recreate a naturally reproducing population in dry years when stream flows reaching the lake are not sufficient for migration from the mouth of the stream. The biggest single factor that limits potential for full recovery of a self-sustaining population is the presence of abundant brook trout in the headwaters. The brook trout will have to be eradicated for complete recovery of ELRT.

Major difficulties faced

- The coordination and dialog among agencies and stakeholder groups to reach agreement about restoration goals, including modifying the Pine Creek weir to allow passage.
- Changing the perception that fish used for restoration purposes will negatively affect the fishery in Eagle Lake and elsewhere.
- Implementing a research program to show successful migration and spawning of ELRT can take place.
- Demonstrating the importance of brook trout eradication and finding a way to implement an eradication program.

PIT tag antennae


Overview of Eagle Lake rainbow trout habitat

Major lessons learned

- Although ELRT have more than 60 years of total dependence on hatcheries, the ELRT still can complete its natural life cycle. However, the fishery will likely to continue to depend on hatchery production indefinitely.
- Success depends on close cooperation among diverse agencies and stakeholders, who agree on common goals. Such cooperation develops slowly and depends on a few individuals from each agency or group to make sure it works.
- Research is essential to demonstrate that proposed management actions can work.
- Involvement of the local Paiute people (Susanville Indian Rancheria) greatly increased the interest in and likelihood of restoration of natural population.
- Complete restoration of a naturally spawning population is likely many years away because of the difficulties of dealing with alien species and long-term drought.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- There is now widespread agreement among agencies and stakeholders that restoration of a natural population is possible, if difficult to implement.
- ELRT spawned successfully at historical spawning grounds, followed by rearing of young.
- Modifications to the weir at the mouth of Pine Creek has allowed for natural passage of fish.
- Successful spawning has been recorded only in a tributary from which alien trout have been eliminated.
- Upstream habitat in the perennial reaches of Pine Creek as been greatly improved.
- Complete success will depend on brook trout eradication from Pine Creek using piscicides, which is difficult, expensive, and unpopular.
- Persistence of ELRT still depends on hatchery production.

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Setting the stage for conservation success: large-scale watershed renovation and re-introduction of cutthroat trout in the Rocky Mountain region of the USA

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Introduction

The cutthroat trout (*Oncorhynchus clarkii*) is native to the Rocky Mountain and coastal areas of the western United States (US) and is classified into as many as 14 subspecies (e.g., Behnke, 1992). Seven major inland subspecies of cutthroat trout historically occupied most accessible cold water environments from Canada to southern New Mexico. However, all subspecies have incurred significant range reductions primarily due to competition and introgression with introduced salmonids, but also from habitat degradation and exploitation (Young, 1995; Shepard *et al.*, 2005; Pritchard & Cowley, 2006). Lahontan (*O. c. henshawi*) and greenback (*O. c. stomias*) cutthroat trout are listed as threatened under the US Endangered Species Act (ESA) and the other inland subspecies have either been petitioned for listing under the ESA or are considered species of concern by state and federal agencies.

We focus on the northern- and southernmost inland subspecies, although considerations are likely similar for all subspecies. Westslope cutthroat trout (WCT, *O. c. lewisi*) were historically the most widespread subspecies - occupying an estimated 90,800 km of streams and rivers throughout the Columbia and Missouri basins headwaters - but the range of genetically

Released westslope cutthroat trout in Cherry Creek - September 2012 pure populations has been reduced by 76% (Shepard *et al.*, 2005). On the east side of the Continental Divide occupied habitat reduction has been even more dramatic, exceeding 95%. WCT were petitioned for listing under ESA in 1997 but determined not warranted for listing in 2003. Rio Grande cutthroat trout (RGCT, *O. c. virginalis*) were historically found in about 10,700 km of habitat in the upper Rio Grande basin of Colorado and New Mexico, however the distribution of genetically pure populations of this more arid climate subspecies has been reduced by 92% (Pritchard & Cowley, 2006; Alves, 2008). This subspecies was petitioned for listing in 1998 and was added to the candidate list in 2008. Both WCT and RGCT are given special status recognizing their conservation need by the states in which they are found and by federal land management agencies (e.g., a species of special concern by Colorado, and a sensitive species by the US Forest Service). As such, range-wide conservation agreements are in place to guide conservation and restoration activities for WCT and RGCT. Priorities include protecting existing populations and establishing new ones (Montana Cutthroat Trout Working Group 2007; RGCT Conservation Team 2009).

Goals

- <u>Goal 1</u>: Develop a project working group that collaboratively defines leadership roles and responsibilities for all aspects of project coordination, planning, implementation, research and monitoring.
- <u>Goal 2</u>: Select a re-introduction site encompassing a large geographic area with high quality and diverse habitats to support a robust cutthroat trout population with diverse life-history strategies able to resist threats such as climate change, catastrophic events, and invasive species.
- <u>Goal 3</u>: Eliminate non-native competitors in the re-introduction site (watershed or portion thereof) through physical and/or chemical renovation, and prevent their recolonization.
- <u>Goal 4</u>: Establish a self-sustaining population of cutthroat trout large enough to withstand environmental and demographic stochasticity and likely to persist over the long-term (>100 years) with little or no human intervention.
- <u>Goal 5</u>: Establish a monitoring strategy, including relevant research partnerships, that evaluates key project aspects and allows adaptive management of all strategies and methods as the project unfolds, and to improve and guide future efforts.
- <u>Goal 6</u>: Provide the public with opportunity to experience the restored cutthroat trout population.

Success Indicators:

- <u>Indicator 1</u>: A functional working group with effective leadership that provides a regular forum for professional discussion, project planning, delegation of duties, risk-benefit analyses, project implementation, monitoring and adaptive management, and dispute resolution.
- <u>Indicator 2</u>: Complete removal of targeted non-native species from the re-introduction site.
- <u>Indicator 3</u>: Establishment of a genetically pure, consistently reproducing (e.g., multiple age classes) cutthroat trout population that persists without chronic management or intervention.

Temporary fish movement barrier on Cherry Creek

- Indicator 4: A robust cutthroat trout population that supports recreational (e.g., angling) use and provides genetically pure gametes and individuals in support of other regional cutthroat trout restoration and re-introduction projects.
- Indicator 5: Monitoring and research insights published in peer reviewed literature and adaptively integrated into project implementation allowing the project to proceed more effectively and inform future restoration efforts.

Project Summary

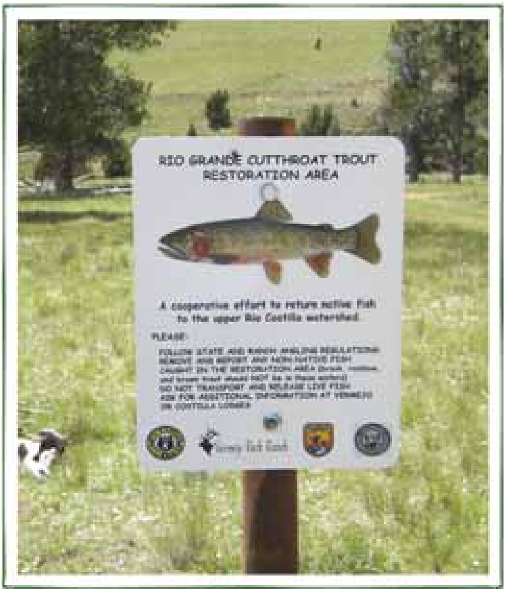
We review case studies of two of the largest cutthroat trout restoration projects ever undertaken in the United States. These case studies embody the goals and challenges inherent in other cutthroat trout recovery projects.

Case Study 1 - Cherry Creek Native WCT Project, Madison River Drainage, Montana (MT): This project encompasses approximately 100 km of stream habitat and 3 hectares of lake habitat suitable for cutthroat trout, and is the largest piscicide renovation project ever completed for the purpose of cutthroat trout conservation. The majority of the project is on private lands and is a collaborative effort among the private land owner - Turner Enterprises, Inc. - and public resource management agencies - MT Fish Wildlife and Parks and the US Forest Service. The Cherry Creek project began with establishment of a collaborative working group, feasibility analyses, and environmental planning in 1997. Opposition to the use of piscicides and non-native fish removal, through a series of legal and administrative challenges, delayed initial piscicide application until 2003. Because of the large spatial scale of the project, the watershed was treated in four "phases", with each phase treated on at least two separate occasions. The piscicide antimycin was applied at a targeted rate of 10 ppb (active ingredient) to remove rainbow (*O. mykiss*), brook (*Salvelinus fontinalis*), and Yellowstone cutthroat (*O. c. bouvieri*, stocked in Cherry Lake in the 1920's) trout from phases 1 and 2. Rotenone (50 ppb a.i.) was used to eliminate the non-native trout in phases 3 and 4. While phases were isolated from recolonization during treatment by natural or temporary man-made fish movement barriers, the entire project area is protected from reinvasion by an 8 m waterfall at the downstream end of phase 4. Piscicide applications were completed in 2010.

In 2006, WCT introductions began in phase 1 via remote stream-side egg incubators and were completed by stocking young of year fish in phase 4 in 2012. Approximately 37,000 eyed eggs and 8,500 young of year fish from multiple wild populations and a hatchery conservation broodstock were introduced. All temporary fish barriers were removed in 2011 to reconnect the phases. Post-treatment monitoring documented WCT throughout the mainstream project area by 2012 and at least two years of natural reproduction, while finding no remaining non-native salmonids. Throughout the project researchers and managers collaborated on project implementation and evaluation, which most efficiently used available resources. Research and monitoring will continue to follow population recovery, comparative survival and fitness of the source stocks, movement into vacant habitats, and impacts to non-target organisms. The Cherry Creek project is a significant conservation achievement for WCT on the east side of the continental divide. This project increases the stream km occupied by WCT in the Madison River basin from 7 km to over 100 km or from 0.3% of historical occupancy to almost 5%. Perhaps more importantly the success of the Cherry Creek project, and lessons learned from, has catalyzed several other cutthroat trout re-introduction projects in southwestern MT. It is important to note that due to the large barrier falls, the Cherry Creek project area was historically fishless. Thus, this project is actually a novel introduction of WCT to a previously inaccessible area within the subspecies historic range.

Case Study 2 - Costilla Creek Native RGCT Project, Rio Grande River Drainage, New Mexico (NM) and Colorado (CO): The most ambitious watershed renovation project ever initiated on behalf of cutthroat trout, the Costilla Creek project encompasses approximately 190 km of stream habitat and 20 lakes. A collaborative effort among Vermejo Park Ranch, NM Department of Game and Fish, CO Parks and Wildlife, US Forest Service, and Trout Unlimited, this project was initially designed to include only 22 km of stream and four lakes protected by a man-made fish migration barrier. Project planning was initiated in 1998 and piscicide (antimycin) was first applied in 2002 to remove non-native brook, rainbow, and brown (*Salmo trutta*) trout from historical RGCT habitat in the headwaters of Costilla Creek. RGCT were re-introduced by stocking 9,500 young of year fish from CO Parks and Wildlife RGCT hatchery broodstock into the renovated stream habitat for three consecutive

Crew member applying piscicide in a tributary to Costilla Creek - August 2011



Costilla Creek project notification

years (2002 - 2004). By 2005 the post-treatment RGCT population was similar in average size and overall abundance to the pre-treatment non-native trout population. Unfortunately during a 2004 lake restocking event, rainbow trout were inadvertently introduced back into the project area.

Administrative and regulatory resistance prevented immediate localized (to stocking sites) retreatment to remove the non-natives and by 2007 hybrid rainbow-RGCT trout were captured. In 2008 a large portion of the project area was successfully retreated with rotenone (50 ppb a.i.) to remove these hybrids. This time, mixed-aged individuals from the NM Department of Game and Fish RGCT hatchery broodstock were introduced (1,900 in 2008 and 10,200 in 2009) and the population recovered by 2010, with no evidence of hybrids or other non-natives remaining. A 2007 environmental assessment proposed expanding the project area to its current size. Watershed renovation is currently ongoing in phases, but the project is complicated due to its size; regulatory requirements; the need for at least seven man-made, temporary fish movement barriers; a 15,700 AF reservoir, and public resistance. To date over 100 km of stream (50% on private land) and 10 lakes have been successfully chemically renovated and restocked with RGCT. If this project is fully completed by 2020 as scheduled it will represent a 20% increase in the amount of stream RGCT currently occupy within their historical range. This project is the flagship restoration effort on behalf of RGCT for the NM Department of Game and Fish. Planning and implementation of the Costilla Project is largely responsible for the development of consistent NM state guidelines regarding the use of piscicides, and for re-development of the Department's native cutthroat trout hatchery broodstock; both important steps for range-wide restoration and conservation of the species.

Major difficulties faced

- Selecting restoration and re-introduction sites of suitable conservation scale, where both landowners and managers will participate, with sufficient habitat quality to allow long term persistence of re-introduced populations.
- Regulatory requirements, administrative processes, and public resistance to the use of fish toxicants (piscicides) requires a significant investment of time, resources, and emotional energy prior to project implementation in the field.
- Locating accessible temporary and permanent barrier sites suitable for designs that are affordable, removable, and functional to keep non-native trout from reinvading the project site during and after piscicide application.

- Finding suitable donor populations for re-introduction when there are few remaining wild sources or hatchery sources that may not meet genetic objectives or withstand removal of individuals or gametes.
- Assurance that all non-native fish have been removed typically requires at least one piscicide application where no mortalities are observed.
- Minimizing the loss of recreational opportunities (e.g. angling or hiking) within the project area, especially on publicly owned reaches of water.

Major lessons learned

- It is critical to establish an effective collaborative project working group that meets regularly to define project goals, assigns organizational responsibilities, conducts project planning, develops annual work plans, handles public outreach, facilitates frank discussion, and flexibility is key.
- Committed project personnel can and need to be consistent and persuasive with agency and organizational administrators to overcome the social, political, and logistical challenges these types of projects inevitably encounter.
- Designing restoration and re-introduction projects within an experimental or research framework provides an opportunity to collect real-time information that can inform ongoing as well as future projects regarding implementation, methodologies, impacts, and population recovery, among other things.
- Removal of non-native fishes and re-introduction of native fish can be successful over relatively large spatial scales and in complex, diverse habitats if implemented methodically across the landscape.

Success of project

Cherry Creek Westslope Cutthroat Trout Project:

Highly Successful	Successful	Partially Successful	Failure
√			

Costilla Creek Rio Grande Cutthroat Trout Project:

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- Implementation of a systematic approach to completely remove non-native competitors over a large scale was instrumental in achieving successful eradication.
- An effective collaborative partnership between private conservation organizations and public resource management agencies created a shared vision, spread financial obligations, and pooled resources.
- Using the best available science and a real-time experimental framework informed project planning, implementation, and cutthroat trout re-introduction

in an adaptive manner and led to improvements and increased efficiency on the project.
- Selection of re-introduction sites with high quality and diverse habitat at a scale appropriate to support all life stages of native cutthroat trout has allowed the re-introduced population to persist.
- Persistence and mutual support of project partners through significant social, political and logistical challenges maintained the cohesion and will to complete the project.
- A conservation minded private landowner willing to withstand the risk of failure.

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Re-introduction of European tree frog in Latvia

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Introduction

European tree frog was considered as extinct in Latvia since last decades of the 20th century. Data on the former distribution of this species are rather incomplete. Several faunists of German origin (Fischer, Seidlitz and Schweder) have mentioned the species as being present in Latvia in the 18th - 19th centuries (Silins & Lamsters,1934). Several reports have even been received in the 1980s (Zvirgzds *et al.*, 1995). Intensive agriculture, rapid deterioration in total area covered mainly by wetlands, and extinction of beaver (*Castor fiber*) in Latvia in the end of 19th century, could be the main factors, which could cause the vanishing of *Hyla arborea* from Latvia. The re-introduction program was started by Riga Zoo in 1987, and a total of 4,110 juveniles in total were released in SW Latvia (Liepaja district), where protected area with total area of 350 ha was established in 1999. The area accommodates a large number of ponds, considerably changed by beavers. Before the re-introduction started, the European tree frog was listed in Red Data Book of Latvia under Category I (endangered species) (Latvijas PSR Sarkana gramata, 1985), at meantime Category II (vulnerable species) (Latvijas Sarkana gramata, 2003). The European tree frog is included in Appendix II of the Bern Convention.

Goals

- Goal 1: Creating sustainable populations of European tree frog in Latvia.
- Goal 2: Proving that creating sustainable populations of amphibians in nature is possible by releasing of specimens, bred under laboratory conditions.
- Goal 3: Proving that *Hyla arborea* can survive Latvia climatic conditions, therefore this species most likely was a natural part of Latvia nature during past centuries.

Success Indicators

- Indicator 1: Self-sustaining population established at re-introduction site, with more than 10 generations developed naturally.

European tree frog © Sergey Cicagov


Typical pond habitat © Andris Eglitis

- Indicator 2: The distribution of the population around the re-introduction site, as far as the suitable environment is available.

Project Summary

Feasibility: Laboratory of Ecology (Amphibian Department since 2006) was founded in Riga Zoo in 1987 with its main task to re-introduce the European tree frog in Latvia. The re-introduction was planned with captive-bred tree frog youngsters in their first year of life.

The considerations were as follows:

- The translocation of a larger amount of adult specimens from other natural populations could place the donor population at risk, even if the population is considerably stable.
- The youngsters would have a considerably higher ability to adapt to wild conditions than adults, if captive bred specimens are released into wild (Dunce & Zvirgzds, 2005).
- The adult specimens for captive breeding were caught in Southern Belarus, near the confluence of Goryn and Pripyat rivers, what is geographically closest stable population (there is also small population in Lithuania).

Implementation: The adults were kept in outdoor terrariums and fed with artificially bred insects as well as meadow sweeps. At the end of October and early November the frogs were placed in wooden boxes, filled with sphagnum, and boxes were kept in refrigerator for hibernation (average temperature 5°C) till the end of January and early February. Later it was found out that an old cellar as a hibernation place is better for the amphibians welfare, despite greater fluctuations of temperatures (from 1°C - 7°C). After hibernation the temperature was raised gradually, and the artificial daylight period gradually lengthened, imitating the day length of the breeding period. The frogs were fed intensively and breeding was stimulated with hormone injections, using Surphagon, a synthetic analogue of Luliberin (produced by Bapex Co., Latvia). During the first year of breeding effort the hormone treatment was given in the beginning of May, in other years during the beginning of March. In both cases the results were virtually identical.

Two males and one female were usually placed in a 35 liter aquarium with a water level of about 5 cm and several plants. Each female produced 200 - 1,000 or even more eggs. Hatching usually started on the 8th - 10th day of development. The larvae were placed in aquariums with aerated water; temperature was maintained 24°C - 27°C at day, 20°C - 23°C at night. The density of tadpoles never exceeded 2 - 3 larvae per liter. Tadpoles were fed with dried and boiled nettles, meat, aquarium fish food (Tetra) and pollen. The natural photoperiod was simulated using luminescent lamps. The average amount of animals that metamorphosed was 60% - 70% of the initial larvae; in some cases it even exceeded 90% (Zvirgzds *et al.*, 1995). The metamorphosis took 30 - 60 days (in the wild it usually takes 90 days). Froglets were fed with meadow sweeps and captive bred insects. About 2 - 6 weeks after metamorphosis the froglets were taken to the re-introduction site.

Amphibian experts at a potential release site in Latvia during 2004 © Elvira Hrscenovica

During 1988 - 1992 a total number of 4,110 juveniles, progeny from 14 - 17 breeding pairs, were released. All releases were conducted in one locality, enabling accurate further monitoring of population dispersal.

Post-release monitoring: The release site was chosen in SW Latvia (Liepaja district, ca. 56°30' N 21°42'E) where a protected area was established with total area of 350 ha. The first vocalizations of adult tree frog male in the re-introduction site were recorded in 1990 - two years since the start of the re-introduction program. This confirms that under particular conditions males can reach sexual maturity in 2 years. The first tadpoles in the wild were found in 1991, at the release site. The first calling males outside the release site were recorded in 1993. Further distribution progressed even faster and up to 2002, tree frogs were recorded already in 110 localities.

The distribution of the newly created population was monitored mainly on the basis of the spring mating calls. All new-recorded localities were registered by GPS and mapped till 2005. The local communities were informed about the project by dispersing booklets, giving lectures in schools, as well as cooperating with media (TV, radio). In later years the area of the population reached the size what made it practically impossible for accurate monitoring and further dispersal of tree frogs is followed up by reports of local people.

Major difficulties faced

- It is difficult to estimate the present size of population because of extended area. Despite of informational work with local people the reports about tree frogs are occasional and do not show the full picture of species occurrence.

Major lessons learned

- Under laboratory conditions the breeding can be effected to happen earlier than in the wild, and the larvae develop faster. Thus, the released froglets have more time to adapt to natural conditions as well as for feeding and growing. We hypothesize that it could result in a much higher survival rate during the first winter.
- Despite that the breeding of tree frogs was stimulated by hormonal injections in all cases, we did not face any problems regarding tadpole or froglet survival or growing rates.

Success of project

Highly Successful	Successful	Partially Successful	Failure
√			

Reason(s) for success/failure:

- After 14 of initiating the re-introduction program, monitoring data showed that total area of population dispersal covered 800 - 900 km^2 (Dunce & Zvirgzds, 2005). As it could be inferred from later reports, it continues to expand.

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Repatriation of eastern indigo snakes to conservation lands in South Alabama, USA

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Introduction

The eastern indigo snake (*Drymarchon couperi*), historically, ranged across the Coastal Plain of the United States from southeastern Georgia, south through Florida, and west across southern Alabama and Mississippi (Conant & Collins, 1998). But, these large predators likely never were abundant and populations of them were vulnerable to habitat loss and fragmentation, mortality associated with increased roads and vehicle traffic, and collection for the pet trade. In 1978, eastern indigo snakes were listed as Threatened under the federal Endangered Species Act, and, more recently, were classified by IUCN as Least Concern. Wild populations of eastern indigo snakes currently are known only from southeastern Georgia and Florida. Here, we describe an attempt to repatriate eastern indigo

Eastern indigo snake

snakes to the Conecuh National Forest, in south-central Alabama. We build on previous repatriation efforts by Speake (1990), who demonstrated how to rear young for release, but who failed to establish breeding populations at 36 sites chosen for repatriation (Hart, 2002). Our project focuses on a single site, where a minimum of 300 one to two-year old snakes will be released. Within the context of repatriating this species to Alabama, we assess the utility of soft releases as a repatriation strategy for large snakes.

Goals

- Goal 1: Assess utility of soft releases for large snakes.
- Goal 2: Compare movement patterns of repatriated snakes to those published for free-ranging individuals in native habitats.
- Goal 3: Compare habitat selection of repatriated snakes to that published for free-ranging individuals in native habitats.
- Goal 4: Establish reproduction of free-ranging snakes at a repatriation site.
- Goal 5: Document spread of a population from the release site.

Success Indicators

- Indicator 1: Reduced home range size and increased home range overlap for soft released individuals relative to those that are hard released.
- Indicator 2: Home range size of repatriated males and females are within range of values published for Georgia source populations.
- Indicator 3: Patterns of habitat selection of repatriated males and females are within range of values published for Georgia source populations.
- Indicator 4: Production of viable offspring from at least one repatriated female.
- Indicator 5: Discovery of at least one unmarked individual captured at a distance from the release site that is longer than the diameter of an average adult male home range.

Project Summary

Feasibility: Three main questions determined whether this project was feasible? i) can we find a reasonable repatriation site?, ii) can we generate stock for repatriation at this site while minimizing effects on source populations?, and iii) can we raise sufficient offspring to generate stock for repatriation?

Implementation

Question 1: We selected the Conecuh National Forest as the release site. The last known occurrence of eastern indigo snakes in this region was made by Neill (1954). The area has received two decades of restoration of longleaf pine forests, transforming the landscape into one that mimics the structural features of old-growth forest. Growing populations of gopher tortoises are present and provide vital refugia for released snakes. The custom of local snake hunters, who used gasoline fumes delivered to the bottom of gopher tortoise burrows to drive out eastern diamondback rattlesnakes, has been outlawed, removing one source of mortality for released indigo snakes. Finally, the site has a similar road density to that of source populations, limiting a second source of mortality to reasonable levels.

Question 2: We established a cooperative effort with the following agencies: Auburn University, Alabama Department of Conservation and Natural Resources, Georgia Department of Natural Resources, The Orianne Society, United States Fish and Wildlife Service, United States Forest Service, and Zoo Atlanta to conform to state and federal laws and to establish consistent funding for the expected 10 year life span of the project. We limited removal from source populations to gravid females that were retained in captivity until each produced a clutch of eggs; this was followed by release of each adult female back to its source population. A maximum of two individuals were used per source site per year to minimize our effect on the demography of the source populations. We required 6 - 8 gravid females per year over a four-year period to produce stock for a Florida breeding facility (to be used for future repatriation projects) and to produce an average of 30 offspring per year for repatriation to Alabama.

Snake release pen

Question 3: Collaboration with Zoo Atlanta was a vital component of the project. This allowed us to raise large cohorts of offspring in controlled environments over two growth years. These offspring were then maintained at Auburn University in outdoor enclosures for 2 - 4 weeks prior to release in the spring. This final stage allowed us to acclimatize snakes to field conditions by giving them access to sun and shade provided by a variety of shelters, as well as training them to seek appropriate live prey.

Post-release monitoring: Initial three cohorts (hatched 2008 - 2010 and released 2010 - 2012) included an average of 20 snakes per year with radio transmitters implanted. Individuals were monitored 3 - 5 times per week during spring and summer and once per week during fall and winter. Locations were recorded by GPS and mapped to GIS layers for the release area. Most of the year, snakes used xeric longleaf pine and mixed pine-hardwood sandhills and the adjacent riparian zones of blackwater creeks and other wetlands. Shelters used included primarily gopher tortoise burrows and stumpholes, but also included armadillo and small mammal burrows and downed woody debris. In winter, snakes remained in upland habitats, where a majority of individuals occupied gopher tortoise burrows. Home ranges of males were larger than home ranges of females. Movement patterns, habitat use and home range sizes observed post-release were similar to descriptions published previously for free-ranging

Eastern indigo snakes in release pen

individuals studied near the source population sites (Hyslop, 2007). Females established relatively small home ranges near the release site regardless of whether they were soft released or hard released. Soft released males had home ranges of similar size to those that were hard released but retained those home ranges near the release site so that soft released males had increased home range overlap with females compared to overlap patterns associated with hard released males. Survival was not significantly different for hard and soft released snakes. Multiple females were observed to spend significant portions of the breeding season cohabiting refugia with males. One young female, captured in 2011 and refitted with a radio-transmitter, was gravid but had unviable eggs. Two additional females captured in 2012 laid eggs that produced 9 offspring. No unmarked individuals have yet been captured. Continued post-release monitoring will be important to the assessment of Indicator 5 and the evaluation of other project goals.

Major difficulties faced

- Novel disease issues for captive snakes (new species of *Fusarium*; developmental anomalies).
- Behavioral differences between snakes retained only in indoor cages and those that experience outdoor cages before release.
- Early escape from pens through underground root channels and mammal burrows.
- Long distance dispersal of males and emigration to private lands.
- Road mortality difficult to prevent, even in areas with low road density and low traffic.

Major lessons learned

- Soft release may be an effective release strategy for minimizing excessive dispersal.
- Females establish stable home ranges more readily than males.
- Soft release improves reproductive opportunities for females.
- Education can help prevent intentional harm and garner community support.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- Home range size and habitat use are comparable to values from source areas.
- Soft release snakes had decreased home range size and dispersal relative to hard release.
- Home range overlap documents opportunities for reproduction by repatriated snakes are improved by use of soft release.
- Successful production of viable offspring was documented for two repatriated females.
- An additional seven years of releases are required to assess whether the population is spreading.

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Experimental translocation (re-inforcement) of the Hermann's tortoise, Var, France

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Introduction

Hermann's tortoise (*Testudo hermanni hermanni*, Gmelin 1789) is one of the most threatened reptiles in Europe. The decline began in 1910s in Western Europe distribution (Italy, France, Spain and some Mediterranean islands) and its future is seriously jeopardized. The subspecies is the only terrestrial chelonian in mainland France. This subspecies has totally disappeared from Eastern Pyrenees around the 1960s - 1970s. Fragmented populations remain in the Var, mainly in the Plain and the Massif of the Maures. Since 20 years conservation measures have been undertaken. Although the total area of distribution of the species remains unchanged, populations are particularly vulnerable due to habitat loss, forest fire, illegal collection and use of heavy machines for agriculture and forestry.

This species is listed on the Appendix II (A) of the CITES and is classified as "almost threatened" on the World Red List and as "Vulnerable" on the national Red List. The Var populations is "in danger" according to the IUCN terminology. Conservation projects (e.g. translocation) should protect and facilitate the reconnection of the twenty reproductive fragments identified (Livoreil, 2009). It might be a suitable tool to re-enforce the most weakened populations living in the "Plaine des Maures" (Var).

Hermann's tortoise

Goals

- Goal 1: Replace tortoises in a favorable new environment rather than release them into their original but degraded landscape or maintaining them in a care center.
- Goal 2: Identification of potential translocation sites in the species' historic range.
- Goal 3: Determining which individuals to release according to sanitary conditions,

genetic profiles and life story. These tortoises are wild animals from rescue operations or those found wounded.

- Goal 4: Determination of the best season of releasing (spring vs. autumn) considering survival and site fidelity.
- Goal 5: Annual monitoring of individuals (both wild and translocated).
- Goal 6: Re-inforcement of populations in protected area with suitable habitat and low predation risk but weakened by forest fires.

Success Indicators

- Indicator 1: Get a national and ministerial agreement for a translocation plan.
- Indicator 2: Establishment of pre- and post-release monitoring program.
- Indicator 3: Measurement of survival, settlement/dispersal and eco-physiological state of the released tortoises; comparison between released and wild tortoises.
- Indicator 4: Self-sustaining populations established at local site.

Project Summary

Feasibility: Despite conservation measures, every year several wild specimens of Hermann's tortoise are displaced from their natural habitat (rescue operations or found wounded). The rescued tortoises are brought back to SOPTOM but cannot be kept indefinitely. After genetics and sanitary tests, wild native tortoises are maintained temporarily in the breeding facilities. The release of individuals may help to restore native population impacted by a fire (Lecq *et al*., submitted) but such events lead to a high mortality rate (50% - 70 %) and populations need several decades to re-establish themselves. However, such actions face several complications such as homing behavior, and is one of the main elements, that could compromise the establishment and the survival of the released tortoises. Conditions of release may improve the chance of success, but experiments are lacking. Within the framework of the conservation Life+ program (2010 - 2014), we aim to evaluate the effect of season (spring vs. autumn) on the release success (dispersal, survival, reproduction etc.). A translocation plan was accepted at the national and ministerial level in 2012.

A feasibility study focused on the biology of the species was undertaken in 2011 (origin of the individuals, genetics, sanitary, knowledge of the native population etc.). An assessment of potential translocation sites was made by evaluating criteria of eligibility according to IUCN recommendations. Two sites located in the area of historic distribution were chosen. First site is in the National Nature Reserve of the Maures' Plain, second is located in a national forest managed by French government. The first site was damaged by a fire in 1979 and was the place of reforestation. The second was impacted by forestry operations during 30 years (1960 - 1990) and a fire in 1990. We evaluated a group of important criteria for the survival of released tortoises. They were developed from the experience of the Hermann's tortoise re-introduction project in Spain (Bertolero *et al.*, 2007). On both sites, previous counting indicates a very low density of native tortoises and effects of predation were unknown. These sites provide suitable habitat for food resource, thermoregulation, water etc. Long-term protection and control of the sites are effective.

Implementation: We did an initial survey in 2012 on the two sites, one year before translocation, in order to collect ecological (spatial movements, habitat use) and physiological data (survival, thermoregulation and sanitary condition). On both release sites, we followed one pool of tortoise from the native population ("resident group") and one from an adjacent population ("control group"). This allows us to control the possible effect of site, and interaction between the released tortoises and the native population (USFWS, 2011). During this initial survey, we noticed on the second site a strong predation pressure; probably a badger (*Meles meles*) which killed a third of the wild radio-tracked tortoises (3/9).

In 2011 - 2012, with growing concerns about the global impact of emerging infectious diseases, an extensive health screening program of wild origin tortoises held in captivity was established. Because *Mycoplasma agassizii* and tortoise herpesvirus are important pathogens (Salinas *et al.*, 2011), tortoises underwent viral, parasitological, morphological and blood screening prior to release. We chose healthy tortoises from Var origin (genetic).

Post-release monitoring: We decided to implement the experimental design only on the first site exempted from predation. We hard-released 12 individuals within a host population during the 2013 spring (after hibernation emergence). Each pool of released tortoises is followed during the two years in order to quantify dispersion, site fidelity, micro-habitat use, survival, reproduction, etc. Qualitative data relying on thermal behavior, body and sanitary condition, physiology are measured (stress, metabolites, etc.) by blood sampling and data loggers. In parallel, similar measurements are done on both groups of tortoises: native and control. The expected outcome of short term successful translocation could be: high level of survival, stable body condition, low baseline stress levels, similar habitat used by native and released tortoises, easy thermoregulation behavior, etc. The success of the action will be evaluated in the mid-term (3 to 15 years after release) and in the long-term by Capture-Mark-Recapture study.

We radio-tracked both translocated in 2013 and resident/control individuals in 2012 and 2013. More than four months after the spring translocation, the tortoises did not exhibit short-term costs (e.g. decrease in body condition); resident individuals did not display any sign of perturbation caused by the introduction of novel individuals. No mortality was noticed, the tortoises did not scatter more than the resident tortoises (no homing behavior). Movement patterns were typical of the species, with males travelling longer daily distances than females. Shell temperature was highly dependent on environmental temperature, and generally higher, suggesting an active thermoregulation behavior. From a conservation perspective, our results are encouraging for the expected settlement of translocated individuals. Although, it has been suggested that soft-release might be preferable over a hard-release to limit dispersal risk (Attum *et al.*, 2011); our results favor a direct, hence simple approach. This specific question will be tested in autumn 2013 by releasing individuals just before hibernation.

Radio-tracking Hermann's tortoise

Major difficulties faced

- Selection of healthy (absence of mycoplasma) and genetically clean individuals of wild origin (from South of France) - few individuals are available after selection.
- Predators - notably the wild boar (*Sus scrofa*) and the badger remain widespread and a very significant threat and are difficult to control.
- Because of the two points mentioned above it is impossible to completely implement the experiment on both sites. As the number of available individuals is restricted after health and genetic selection (twice lower than what was planned), we decided to implement the experiment only in one site.
- No difficulties faced during the first four months of post-release.

Major lessons learned

- Future health screening programs should take into account that Mycoplasma are present in the wild population. We are implementing a parallel sanitary survey in wild populations and we have at present a reliable method for the detection of Mycoplasma, an emerging infectious disease known to impact gopher tortoises in North America, *Testudo graeca* in both wild and captive populations and *Testudo hermanni* in captivity. We do not know the effect of this disease on Hermann's tortoise wild populations.
- All translocated tortoises are from sites more than 10 km of the release site. As a potential consequence, no homing behavior was observed during the first two months of post-release.
- We did not notice any short-term effects associated to the translocation (e.g. decrease in body condition or mortality).
- Careful field monitoring and the measurement of both movement and ecophysiological parameters help to better evaluate the translocation success in a short term period.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- Tortoises were able to find appropriate micro-habitats and adequately adjusted their thermoregulation strategy to local conditions.
- The short-term success of the experience was possibly influenced by the high quality of the habitat, particularly heterogeneous with abundant natural resources.
- Resident individuals did not show any sign of perturbation caused by the introduction of novel individuals. Instead, we observed mating between resident and translocated tortoises, several females were observed laying.
- Long term monitoring is required to better assess the establishment of the individuals in the resident population.

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Re-introduction of Hungarian meadow viper in Hungary

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Introduction

Hungarian meadow viper (*Vipera ursinii rakosiensis* Méhely, 1893) is an inhabitant of steppe remnants. Recent populations occur on grasslands formed by a mosaic of drying marsh-meadows and sandy pastures, where the relatively diverse features of terrain and grass cover provides high prey-abundance and several different microclimatic options. Recent populations only occur on two places in Hungary: two populations in Hanság and less than 10 in Kiskunság. The total population was estimated under 500 individuals. Hungarian meadow viper is protected in Hungary since 1974, strictly protected since 1988, and was raised to the highest conservation category since 1992, with a conservation value of 1,000,000 Ft (approx. US$ 4,660). Its critical situation was recognized internationally as well, as it had been included in Bern Convention Appendix II, it is listed in CITES Annex I and IUCN categorized as 'threatened' The Bern Convention approved a European Action Plan on *Vipera ursinii* in 2005. The species is listed in Annex II of the Habitats Directive therefore all occurrences were included into Natura 2000 Network. A Species Conservation Plan was approved in 2004 in Hungary and complex conservation project was started co-funded by European Commission's LIFE and LIFE+ funds.

Goals

- Goal 1: Cover locations of all surviving populations and describe them, estimating their size, demography and describing genetic background, habitat characteristics and local threats. All these information were included in the Species Conservation Plan.
- Goal 2: Secure long-term survival of the species on known habitats, by applying

Hungarian meadow viper

Juveniles marked for outdoor terrarium release

appropriate management and increase of suitable habitats through land-purchase and grassland reconstruction.

- Goal 3: Start captive breeding of the species in 2004 with the aim of future re-introduction and population re-inforcement. Techniques were described in the so called Breeding Protocol. The aim was to build up a breeding stock of several generations through successful breeding, which will reliably provide annual needs of re-introduction effort.
- Goal 4: At the start of the re-introduction in 2010, the so called Reintroduction Protocol set a target of altogether 400 released vipers on two sites over a four year period on reconstructed grassland in Kiskunság. The possible release in Hanság by the end of the period, depending on the state of the reconstructed grassland by that time.
- Goal 5: Learn information on the fate of the released individuals with continuous monitoring on release sites, through regular surveys and use of remote sensing tracking.

Success Indicators

- Indicator 1: Size of known viper habitats and spatial information on their management.
- Indicator 2: Size of increase and reconnection of fragmented viper habitats with grassland reconstruction on hills, providing hibernation sites safe from high water-table in winter.
- Indicator 3: Number of vipers born in successful breeding of multiple generations following pedigree and genetic screening.
- Indicator 4: Number of individuals released per site or over years.
- Indicator 5: Number of re-introduced or reinforced populations with estimated surviving number of released individuals. Successful overwintering, breeding and recruitment can be considered as milestones and final proof of conservation effort's success.

Project Summary

Feasibility: The severe decline of the species was mainly caused by habitat-loss. Previously unpredictable water movements were diminished by building of drainage canals, making those sites suitable for agricultural cultivation, meaning ploughing of most of the grasslands. Remaining grasslands were mowed intensively, which was intolerable for the species. Collection for trade purposes and intentional killings further reduced its numbers. Remaining small and fragmented populations became vulnerable and small, local catastrophes could fully destroy them (Nilson & Andrén, 2001; Újváry *et al.*, 2001). A Population and Habitat Viability Assessment organized by IUCN Captive Breeding Specialist Group and Budapest Zoo in 2001, the approved Species Conservation Plan (Dankovics *et al.*, 2004) and the European Action Plan (Edgar & Bird, 2005) all came to the conclusion that complex conservation effort is needed with inclusion of additional elements to ongoing conservation measures, like habitat reconstruction and enlargement and captive breeding and re-introduction.

Implementation: In 2004 the systematic and conceptual conservation program, running since 1993, has opened a new chapter in the story of Hungarian meadow viper with the start of a four-year LIFE-project named as "Establishing the background of saving the Hungarian meadow viper (*Vipera ursinii rakosiensis*) from extinction", led by Hungarian Ornithological and Nature Conservation Society (MME BirdLife Hungary) with participation of Directorate of Kiskunság National Park (KNPI) and Directorate of Duna-Ipoly National Park (DINPI). The conservation effort was secured for the period between 2009 and 2013 thanks to funding by LIFE+ Fund. Beneficiaries of the "Conservation of Hungarian meadow viper (*Vipera ursinii rakosiensis*) in the Carpathian-basin" are MME, KNPI, Directorate of Fertő-Hanság National Park (FHNPI), Budapest Zoo (FÁNK), Compound Eye Film (T.HU), and from Austria Nationalpark Neusiedler-see und Seewinkel (NNSS), Schönbrunn Zoo Vienna (TSV) and Research Institute of Wildlife Ecology, Vienna (FIWI). The set of actions implemented in the conservation program can be categorized into four major groups: monitoring of the species and its habitats; defragmentation and enlargement of recent habitats through grassland reconstruction; captive breeding and re-introduction of the species; information of the public and public awareness campaign (Halpern, 2007).

The Hungarian Meadow Viper Conservation Centre was created in 2004 on a remote farmhouse in Kiskunság. The breeding of the snakes was started with 10 animals, which were collected from four different habitats of Kiskunság in 2004. During 2007 - 2008 we captured a further six snakes, representing other populations, including the two in Hanság. Vipers were placed in pairs or breeding groups in outdoor terraria, providing semi-natural conditions, each equipped with artificial burrows. These burrows were developed in the program, in order to provide safe hiding place and winter hibernacula for the vipers.

Through successful reproduction in each year since the start of captive breeding, until 2012 overall 1,392 vipers were born. Young vipers born at the Centre - thanks to prey-abundance and lack of predators - are reaching maturity in higher percentage than those in natural populations. Until 2008, newborn vipers spent their first winter in separate indoor terrariums, with continuous feeding. On average an annual mortality of 10% was observed in these cohorts. Since 2008, when we were able to include captive raised individuals in the breeding, number of offspring increased significantly. Meanwhile the testing of artificial burrows ensured us about its safe use for wintering of juveniles, therefore since the 2009


Meadow viper in natural habitat

cohort the juveniles are also wintering in outdoor enclosures. In these cohorts a higher first year mortality was observed (20% - 30%), which was compensated by the increase of breeding pairs, in order to ensure the possible release of 100 - 150 vipers of each cohorts when they reach the age of 3 - 4 years.

The first re-introduction took place in 2010, when 30 vipers were released to reconstructed grassland in Kiskunság, in the vicinity of the breeding centre. During three re-introductions altogether, a total of 142 vipers were released to this site and another 45 to another site nearby in 2012. Artificial burrows were used as mediums for release, in order to provide the vipers safe and known hiding places, and a chance for a step-by-step discovery of their new home. Timing of the release was chosen with similar aim, speculating as after the end of winter, vipers generally spend time basking close to their burrow.

Post-release monitoring: Regular surveys were limited to once a week, in order to minimize disturbance. During 2010 and 2011, about 50 - 55 vipers were spotted using different methods. Most of the vipers were seen close to artificial burrows, checked visually by using a pipe-camera. Vipers spotted were not handled, just photos was taken for identification purpose. There were nine individuals identified in 2010 and further eight vipers in 2011, with observations of gravid females and births. The last released group contained six vipers that were previously implanted with VHF-tags, enabling their tracking over a period of 6 - 8 months. To monitor predator presence camera-traps were used in 2012.

Major difficulties faced

- Lack of information about optimal breeding conditions for the species.
- Grassland reconstruction was delayed significantly as removal of planted forests had to overcome many bureaucratic obstacles.
- Effective ways of removal of invasive and alien plant species (*Robinia pseudoacacia*, *Solidago canadensis* & *Pinus nigra*) had to be tested first.

- Small enough VHF-tags with long enough battery life were needed for tracking vipers. After many tested solutions, finally a technician at FIWI developed the ones we are using now.
- High densities of possible predators like wild-boar, fox and badger.

Baiting of rodent traps

Major lessons learned

- Grassland reconstruction seems simple when planning and proved really complex and difficult in the implementation phase.
- Despite some fears of genetic problems, Hungarian meadow vipers can reproduce annually with an average clutch size of 11, with a record of 27.
- Artificial burrows can be useful tool in providing semi-natural conditions and chance for regular checks. Even they can be used in translocating animals to new sites.
- Post-release monitoring needs reliable remote tracking technique. Camera-traps can provide additional information on presence of predators.
- Although the target species is a venomous creature, general public is neutral or supporting the project.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- The complex approach of conservation effort tried to tackle each element that might be responsible for the detected decline of the target species, involving all stake-holders, NGOs, National Parks and state authorities who have connection to the subject. This effort was awarded by the European Commission by naming the LIFE-project as “Best of the Best” in 2009.
- Captive breeding technique of Hungarian meadow vipers evolved during the project to a level, that planning of any current or future repatriation can rely on.
- Habitat reconstruction effort and recent changes in management of viper habitats influenced positively overall state of remaining viper inhabited sites.
- It would be too early to claim re-introduction effort totally successful, but there are positive signs like observed reproduction in the wild. Hopefully in a few

years time we will have more proof of success in this field, and we can claim the project “Highly successful”.

- Public opinion is rather positive about the conservation effort, thanks to careful but widespread communication of project goals and results, e.g.. the regularly updated website of the project: www.rakosivipera.hu/en/

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Re-introduction of vinaceous Amazon parrots in the state of Sao Paulo, Brazil

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Introduction

The vinaceous Amazon (*Amazona vinacea*) is an endemic species of the Atlantic rainforest and submontane mixed regions. In the past it was widespread through eastern South America, and now isolated in small islands of habitat due to heavy deforestation coupled with capture for the illegal trade. An approximate number of less than 2,000 individuals remain in Brazil, and populations in several states are close to extinction (Birdlife International, 2013). The species is classified as CITES I and listed globally Endangered by the IUCN as well as nationally and critically endangered in the state of Sao Paulo (Birdlife International, 2013; Livro Vermelho, 2008). From July 2011 to the present moment groups of vinaceous Amazons were selected to participate in a re-introduction in a private protected area in the Atlantic Rainforest, at the state of Sao Paulo, Southeast Brazil. In this area the species had been declared extinct for at least 30 years and an effort for re-establishing it in its historical range is being carried on. Birds were chosen according to their ability to fly, behavior, physical and health screening. All were confiscated and previously maintained as illegal pets having undetermined ages, although considered as adults based on their sexual behavior.

Goals

- Goal 1: Re-introduction of the species in its historical range.
- Goal 2: Gradual adaptation of individuals that would remain long enough around the release area receiving supplemental food in order to be self sustainable in the wild.
- Goal 3: Birds forming a local population and remaining in the area long enough to form flocks before migrating to other regions of the Atlantic Rainforest.

Released bird in the wild © Wallace & Wittkoff

Pair inspecting a nestbox © Wallace & Wittkoff

- Goal 4: Formation of pairs and breeding attempts.

Success Indicators

- Indicator 1: Survival of most released individuals forming an independent flock.
- Indicator 2: Use of wild food sources demonstrating not being dependent of supplemental feeding.
- Indicator 3: Proof of successful fledging of offsprings in the wild.

Project Summary

Feasibility: The chosen region for the re-introduction is located inside the Lymington Foundation a 36 ha property of protected restored habitat which promotes captive breeding of endangered parrot species as well as conservation initiatives at this area and elsewhere. The Foundation area is also located within the original distribution range for the species. The surrounding areas are not subjected to agricultural or increased human expansion pressures and a good relationship with the neighboring communities is maintained.

Implementation: Birds were subjected to a quarantine period and health exams and later gradually installed in groups in a intermediate sized flight (3 m^2) in order to get used to the temperature, sights and sounds of the area as well as to observe their behavior (pair formation and antagonistic interactions that would need intervention). Once observed they were adapted and passed through a new series of health checks, the groups were installed in a large suspended flight (12 m long) for flight training and presentation of wild food types found in the region. Branches of local trees with fruits and pods with varying diameters and with their respective leaves are offered frequently and set up in a way to force birds to fly to them and use as perches. This is especially important for heavier birds of Amazon parrot size and up to learn how to select an appropriate branch that will support their weight in order to land safely and how to reach food in thin/hard to reach braches.

Observation of each individual's behavior, weight and breast muscle development checking allowed selection and release groups of 3 to 4 birds each time for easier follow up instead of releasing many birds at once which could create some difficulties on visual monitoring. These birds are closed in a 3 m section of the flight for a few days before opening the release hatch in the early morning offering food at this platform as well on elevated feeders set in front of the flight path exit.

Release flight and habitat © Andre & Saidenberg

The door is closed at night (to avoid the entrance of predators) with any birds that wish to come back to sleep inside and opened the next morning. When the group seems to be adapted, another is prepared also taking into account the climate conditions (not heavy rainfalls or release in a time where wild food sources might be scarce such as winter in this region). Artificial nestboxes were also set up around the release area and immediately called the bird's attention.

Post-release monitoring: Visual monitoring is done daily especially during feeding times (morning and late afternoon) and birds are usually first located by vocalization. More intensive searches are performed especially on the third day if a new bird has not returned to feed. Since the ex-pet background creates some additional difficulties on some individual's adaptation, any bird that seems to have difficulty to come back is offered food by the means of a mobile feeder located close to the tree where it stays or if necessary the bird is captured and brought back to the release area until it gets used to the surroundings and know when to come back for supplemental feeding if necessary. A fact which usually does not have to be repeated more than twice. Since distinctive color markings cannot be easily used for this species to identify each bird individually (e.g. feather color combinations) every individual can be identified by color marked with imping of a central tail feather from a different species (e.g. golden conure, white swan, etc.) as well as with stainless steel leg band (although of difficult identification from distance). Anodized aluminum colored bands were employed recently in a number of birds with success. Monitoring includes the immediate vicinity of the property as well as reports from neighbors and inhabitants of the closest village (7 km away). After one and a half year post release (June 2012), 16 out of 21 released birds could be accounted at the area on certain days although they have been seen more and more infrequently. The individuals have either joined a large flock or stay in small groups visiting the release area only occasionally and being seen eating wild food sources with no dependency on the supplemental feeders. Breeding activity (copulation) was frequently observed with three pairs during the start of the breeding season (September until March) and one pair laid three eggs in an artificial nestbox with one embryo not developing more than one week, one broken, and one infertile. Another pair laid fertile eggs in a dead palm tree hollow successfully raising 3 fledglings.

Major difficulties faced

- Intensive pre-release preparation as well as the need to evaluate each bird's necessities considering their ex-pet background requiring intensive follow up post release to intervene if necessary to guarantee a high percentage of survivability.
- Keeping track of newly released birds on heavily forested and steep terrain with no available open tracks.
- Territorial aggression by established birds toward new candidates during breeding season required to temporarily recapture of a few previously released birds until the new ones had adapted and could be part of the flock.
- Political obstacles created by colleagues and groups who disregard previous re-introduction examples all around the world and consider it as a "novelty" and therefore impractical carrying too many risks.
- Interference by other species such as invasion of nestboxes by Africanized bees and wasps requiring removal and use of safe insecticides. Nocturnal predation of at least one parrot by a big-eared opossum (*Didelphis aurita*), and the presence of crab eating foxes (*Cerdocyon thous*) around the area predating other free ranging birds in the property during the day. These latter species currently lack their own predators for population control due past extinctions and have become extremely common as the new top predators.

Major lessons learned

- For some birds it was found useful to temporarily limit the complete flight ability to fly too far from the release area in the first days where some might get disoriented by plucking two inner flight feathers from each wing which still leaves plenty of flight ability. These grow back within a month when the bird is fully adapted to the surroundings.
- Necessity to prevent access and control common predators by the means of adding plastic sacs wrapped around tree trunks and supplemental feeder's poles covered with grease to prevent climbing predators.
- Importance of habituating parrots in captivity not to perch or forage close to the ground (high perches and suspended feeders in the acclimation flight).
- Capturing and relocation of these common predators.
- Necessity of educating local people to the importance of the release and getting their cooperation.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- Successful establishment of the species in the area with reported breeding attempts as well as fledging of offsprings.
- Careful pre release preparation (flight training, health checks, and wild food presentation) as well as attention on each individual's necessities (removing birds from the main flight that are not being able to compete with stronger/

more aggressive ones, and re evaluation to add them back to the main flight at a later stage).

- Guaranteeing that for the critical first 3 days after the release the birds are able to have access to food and water and learn their way from the property area back to the feeders.
- Attention to predator control.
- Training in captivity to be able to be competent in flight abilities, recognizing wild food types as well as not to look for food at the ground level.

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Re-introduction of crested ibis on Sado Island, Japan

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Introduction

The crested ibis (*Nipponia nippon*), is endemic to east Asia and was historically widespread from Russia, China, Korea, Taiwan and Japan. The Japanese population went finally extinct in the wild in 1981, when last 5 birds were captured on Sado Island and were brought into a captive breeding program. Sado Island has 855 km^2 in area, and located at 40 km offshore from Honshu mainland, where is situated at 280 km north of Tokyo. Seven birds were rediscovered in Shaanxi Province, China, in 1981 after extinction of Japanese population. The crested ibis is listed as Endangered in IUCN Red List and is in CITES Appendix I. The ibis has been protected as a special natural treasure since 1952, and has been protected as national endangered species since 1993 by Japanese Government. The captive breeding attempt from Japan origin birds ended in failure when the last bird died in 2003. However, the program has been sustained using birds brought from China, and successful captive breeding has enabled implementation of plans to re-introduce the species to the wild. In order to re-establish a wild population a re-introduction has conducted on Sado Island since September 2008.

Crested Ibis (M67) foraging a loach in a set-aside paddy

Goals

- Main Goal: To establish a self-sustained population of crested ibis coexisting with human beings on Sado Island.
- In order to achieve the main goal of re-introduction of crested ibis on Sado Island the followings steps are important:

A foraging Ibis (F03) and a working farmer (April 2011 before planting rice)

- Sub-Goal 1: To keep a captive population of ca. 220 ibises with high genetic diversity for ensuring re-introduced individuals.
- Sub-Goal 2: To restore the ecological food web in which crested ibis are positioned as higher predators and umbrella species, by reducing pesticide or chemical fertilizer in the agricultural ecosystem.
- Sub-Goal 3: To maintain 'Satoyama' landscape which will be characterized by sustainable natural resource management based on the benefits of living in harmony with nature. This kind of landscape is preferable habitat for the crested ibis.

Success Indicators

- Indicator 1: Released individual should be settled on Sado Island, and more than 60% of individuals should survive until the following year.
- Indicator 2: Mean annual survival rate of adult birds should be better than 71%, which is identical to the Chinese wild population.
- Indicator 3: Re-introduced birds should produce viable offspring. Reproductive success should exceed 57%, and it will be improved to 67%, which is identical to the Chinese wild population.
- Indicator 4: Re-established population should maintain more than 60 individuals on Sado Island, which should include viable offspring.
- Indicator 5: Re-introduced population should be self-sustaining and show gradual increase without releasing any additional birds.

Project Summary

Feasibility: The captive breeding program using Japanese origin birds ended in failure when the last bird died in 2003. The captive breeding program has been now sustained using birds brought to Japan from China since 1999. Japanese and Chinese populations of the crested ibis have been confirmed to almost identical by comparing whole mt-DNA sequences (Yamamoto, 2009). The last crested ibis inhabited mountain area before extinction (Yamashina & Nakanishi, 1983). The previous range is protected as 'Ko-sado' National Wildlife Reserve, which is 12,620 ha in area. The ibis foraged loaches, frogs, and invertebrates at terraced paddies and small streams, and nested mainly on pine trees (Yamashina & Nakanishi, 1983). Terraced paddies have been abandoned by population decrease and population ageing on Sado Island because there is insufficient

Fishway system installed at a paddy in Kuninaka Plain - person with a straw hat is one of the authors - Dr. S. Yamagishi

farmland for rice cultivation. Streams were modified and were covered in concrete. Moreover, 90% of pine trees were dead because of spreading pine wilt disease. Japanese marten (*Martes melampus melampus*) has been introduced from the mainland to control hare population on Sado Island in 1950s, which became a potential predator for the ibis. Then, foraging and breeding habitats for the ibis have drastically changed over the last 30 years. People have legends that the ibis was harmful for rice cultivation because of trampling on shoots just after rice planting. Japan and local governments repeatedly gave local people environmental education to build a consensus with farmers before re-introduction.

Implementation: In order to re-establish a wild population, the re-introduction program of the crested ibis was announced by the Japanese Government, and it has been implemented by The ministry of Environment, Japan, since 2005. The ministry has established a Re-introduction Center for Crested Ibis equipped with a large training aviary since 2007. In order to increase prey species, the local government of Sado City, also encourages farmers to cultivate organic rice by reducing pesticides and/or chemical fertilizers. The local government has established a certification system for organic rice which will require more than 50% reduction of pesticide and chemical fertilizer use with biodiversity enhanced activities. These activities are installation of fishways, arranging biotopes, making water channels within paddies, or flooding in winter. First release of the ibis commenced on 25th September 2008, and 10 birds were 'hard' released, when the captive population exceed 100 birds. In order to minimize conflict with farmers, the first release was conducted after the harvesting of rice in fall. Any pair bonds did not establish in 2009 because only males settled on Sado Island and all females dispersed to Honshu mainland. To promote quick settlement and flocking, birds have been 'soft' released since 2009. So far, 107 birds were re-introduced in 7 releases between 2008 and 2012. All re-introduced individuals came from captive-bred stocks, which is maintaining almost 200 birds and is scattered in six facilities within Japan to minimize risk of infection.

Post-release monitoring: One-third of released birds were fitted with solar-powered Argos/GPS PTTs and locations are recorded every 3 hours in the daytime. All individuals were also tagged with a bio-tip, uniquely color-ringed, and banded with numbered rings. All birds staying on Sado Island were detected by intensive monitoring after the releases, even though PTTs were not fitted. Information of birds detected by local residents has been also accumulated through public phone call or web sites. Information of birds dispersed to the Honshu mainland was also gathered through local branches of the Ministry of Environment. Only four dead ibis were found and several birds disappeared shortly after the release. We operationally treated birds as dead in the wild if there were no sighting records for longer than 12 months. Nest locations, breeding schedules, individuals involved and nesting success have been recorded for all nesting attempts since 2010. The causes of nesting failure have also been recorded, if identified. Only eight chicks out of three nests have fledged in 2012, so far.

Major difficulties faced

- Released individuals from the captive reared population showed a low reproductive success. Hatching failure or infertility might be major causes of low reproductive success.
- Many nests suffered from predation by crows and martens. Nine birds were also killed by a marten which entered a large aviary while training for the third release in 2010. Females fitted with PTTs were attacked by goshawks only in severe winter. Predation pressure might be high in the wild.
- Genetic diversity of captive population is almost a half of Chinese one because captive population was established from only 5 founders which came from China. It is sometimes difficult to get new founders with novel genetic characters due to recent political issues between Japan and China.

Major lessons learned

- “Hard” release induced females to further breeding dispersal outside the island and ‘soft’ release appears to encourage birds to remain near the release site and to form a flock immediately after release. The number of females dispersed to the outside is decreasing as the number of birds settled in Sado Island increased.
- Adult crested ibises showed high annual survival rate (>0.7), if they can survive for 6 month after release (Nagata & Yamagishi, 2011). Birds surviving in the release event suffered higher mortality in summer than in winter.
- The crested ibis rely on paddies and surroundings as a foraging habitat (Endo & Nagata, in press). The ibis, however, cannot use paddies itself in summer, when rice plants were grown up. Though loaches (*Misgurnus anguillicaudatus*) were predominant prey species throughout the year, earthworms and large insects become important prey items in summer (Endo & Nagata, 2012).
- The ibis can use any tree species even if they can put nesting materials on the fork of tree. This means nesting tree is not restricting factor of successful re-introduction for the crested ibis. Hand reared individuals, however, showed lower mating success, and some cannot complete a nest.
- Crested ibis preferred agricultural ecosystems in lowland areas, so-called ‘Satoyama’, that have been modified over a long history of interactions between human and nature, to those in mountain areas.

Success of project

Highly Successful	Successful	Partially Successful	Failure
		√	

Reasons for success/failure:

- Recently, almost the released birds tended to settle on Sado Island, as population increased.
- Adults of crested ibises showed moderate survival rate (61%) for the first year, and showed higher annual survival rate (~80%) after the second year from the release (Nagata & Yamagishi, 2011).
- Though eight young of three nests fledged in 2012, the breeding success is considerably lower than those of Chinese wild populations (Ding, 2004). We do not know whether they are viable offspring or not, as it will take another two years for young to reach breeding age.
- Current breeding performance is not enough to maintain the population without the release of birds. Though captive reared individuals might show low breeding success, it will gradually impove as offspring born in the wild will increase.
- The effects of low genetic diversity and/or inbreeding depression are still unknown.

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Re-introduction of brown-headed nuthatch & eastern bluebird to South Florida pine rocklands, USA

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Introduction

The Eastern bluebird (*Sialia sialis*) inhabits a variety of open forest types, both deciduous and coniferous, in eastern North America. The brown-headed nuthatch (*Sitta pusilla*), a cooperative breeder, is endemic to pine forests in the southeastern United States. Both species remain common and widespread in appropriate habitat and are not included on any regional or national conservation lists. They are listed by the IUCN as species of Least Concern. Although both species require cavities for nesting, their breeding and foraging ecologies differ. The bluebird relies on existing cavities, while the nuthatch is capable of excavation. Bluebirds forage on a variety of ground-dwelling insects and during the winter consume small fruits. The nuthatch's diet is dominated by pine seeds, which it caches, and insects gleaned from the trunk and branches of trees. Both species, along with three other cavity-nesting bird species, were extirpated from Everglades National Park (EVER), Florida, USA following large-scale habitat loss and degradation of the pine rockland (slash pine; *Pinus elliottii* var. *densa*) ecosystem. The re-introduction of these two species was viewed as a test of the progress made in the recovery of this fire-dependent ecosystem from logging and the implementation of a natural fire-management program.

Goals

- Goal 1: Develop and implement translocation strategies for Eastern bluebird and brown-headed nuthatch.
- Goal 2: Monitor reproduction and survival rates in the re-introduced population to evaluate translocation methods and re-introduction success.
  Goal 3: Establish viable breeding populations in EVER with a population size of >200 breeding

Eastern bluebird © Skip Snow

Brown-headed nuthatch © Gary Slater

territories for each species.

Success Indicators

- Indicator 1: Short-term: Released individuals and their offspring breed successfully.
- Indicator 2: Short-term: Population size increases annually.
- Indicator 3: Short-term: Demographic measures (reproduction and survival) in the re-introduced population are similar to a high-quality reference population (i.e., the donor population).
- Indicator 4: Long-term: Populations maintain a growth rate >1.0.

Project Summary

Feasibility: A qualitative feasibility assessment indicated that pine-rockland habitat in EVER could support breeding populations of Eastern bluebirds and brown-headed nuthatches and that without re-introduction these species were unlikely to recolonize on their own (Slater, 1997). Vegetation characteristics of the forest in EVER appeared comparable to those where large populations of nuthatches and bluebirds thrived. In addition, snags, which often limit cavity-nesters, were unusually abundant after Hurricane Andrew in 1992. The estimated carrying capacity of EVER forests was >200 breeding pairs of each species.

Implementation: We conducted translocations from 1998 - 2001 (Slater, 2001). We captured all nuthatches and most (76%) bluebirds from nearby source populations in Big Cypress National Preserve, approximately 40 km from the re-introduction site; remaining bluebirds were captured from golf courses in Naples, Florida, approximately 140 km away. Translocations of nuthatches, which maintain year-round territories, were conducted from November to February to avoid the peak of breeding activity, ensuring sufficient time to excavate nest cavities. Because they are monomorphic and cooperative breeders, we only conducted translocations when the entire group was captured. Nuthatches were placed in small (1 m x 1 m x 2 m) aviaries and held for 1 - 10 days. In Year 2, we attempted to use large (2 m x 2 m x 2 m) aviaries; however, several nuthatches died and we returned to original methods. For bluebirds, we captured pairs as they established territories (March - April), although we moved a few pair with dependent young (~10 days old) later in the breeding season. Bluebird pairs were placed in large aviaries and held for 1 - 3 weeks, except two pairs that nested and remained in captivity until the young fledged. We released bluebird pairs with nestlings once the young had fledged and were capable of sustained flight.


Pine rockland habitat
© Lauren MacDade

Aviaries were constructed to be mobile and permit open views, while providing protection from the elements. They contained multiple perches, a nestbox for roosting, and food (mealworms, crickets) and water. Release sites were selected based on the presence of suitable habitat, and upon establishment releases were conducted adjacent to occupied territories.

We captured and translocated 53 nuthatches. Six individuals died in the aviary, all within 24 hours, and 5 individuals were released when their condition appeared to deteriorate. Of the 42 released in good condition, 25 (60%) established a breeding territory. We captured, translocated, and released 47 eastern bluebird adults: 17 breeding pairs, 1 single female, and 6 pairs with dependent young. Overall, 31 of 47 (66%) adults established a breeding territory. Five of 18 juveniles were killed inside the aviary by snakes or other predators, presumably crows, that attacked them through the hardware cloth. Only one juvenile released established a territory in the subsequent year.

Post-release monitoring: We met our short-term indicators of success. We found evidence of breeding in each year, with both translocated individuals and locally-produced offspring reproducing successfully. Population size increased in each year of the translocation period, reaching 31 and 38 adults, respectively, for nuthatches and bluebirds. Reproduction and survival estimates were either higher (reproduction) or did not differ (survival) in the re-introduced populations compared to a high-quality reference populations during the translocation period and two-year post-translocation period (Slater, 2003).

We conducted post-translocation monitoring from 2002 - 2003 and 2005 - 2007 for nuthatches and through 2009 for bluebirds. Success, based on population growth rate estimates, following the cessation of translocations was mixed. Following translocations, annual counts of adult nuthatches increased dramatically, reaching a high of 87 adults in 2005, but then declined to 52 adults in the following two years. Reverse-time capture-recapture models found population growth rate estimates were, on average, >1.0, although estimates varied annually (Lloyd *et al.*, 2009). Models indicated that population declines in 2006 - 2007 were due to low survival. We suspect that the effect of two hurricanes in the fall of 2005 may have reduced food availability in subsequent winters by stripping pine trees of their cones. Qualitative monitoring after 2007 indicated that nuthatch population size and distribution increased in subsequent years.

Bluebird populations following translocations increased to 46 adults in 2002, but then declined and varied from 34 to 39 adults until 2009. Population models yielded a population growth rate estimate of 0.92 (95% CI = 0.83 - 1.00), indicating we failed to meet our criteria of success. Monitoring revealed two potential factors stemming population growth. First, bluebird fecundity declined substantially after translocations were discontinued, apparently from high levels of predation, possibly mediated by declining cavity availability. Second, bluebirds, especially juveniles, appeared vulnerable to mortality via vehicle collisions due to their propensity to nest and forage along roadsides. Efforts to identify specific limiting factors in 2008 - 2009 were unsuccessful and we failed to obtain additional funding for further research.

Major difficulties faced

- Successfully capturing the complete pair (bluebirds and nuthatches) or cooperative group (nuthatch).
- Nuthatches mortality in the aviary due to stress inside the aviaries, particularly for those individuals that were captured closer to the breeding season.
- Vulnerability of bluebird nestlings to predation inside the aviary, both from predators that gained entry into the aviary and those external to the aviary.
- Increasing predation rates of bluebird nests after the cessation of translocations.
- Identifying limiting factors hindering population growth of eastern bluebirds in the re-introduced population.

Major lessons learned

- The use of a lure bird to capture bluebirds increased capture success dramatically.
- Translocation success increased as population size increased for nuthatches; in the final two years of translocations, success increased to 73%.
- Most nuthatch groups and bluebird pairs did not maintain pair bonds following release, indicating that future translocations may not need to focus on capturing established breeding pairs.
- In the case of nuthatches, a relatively small number of translocated birds (42) was sufficient to establish a population.
- Long-term monitoring (>10 years) is required to fully evaluate re-introduction success, and long-term funding is required to address unforeseen problems and address causes for why re-introductions fail.

Success of project

Brown-headed Nuthatch:

Highly Successful	Successful	Partially Successful	Failure
√			

Reason(s) for success/failure:

- The development of effective release strategies.

- High rates of reproduction and survival leading to positive population growth rate, acknowledging the population did not reach the estimated carrying capacity during our monitoring period.
- Population resiliency, which allowed the population to recover following a strong decline.
- The nuthatch's ability to excavate its own cavities, particularly in snags of smaller size that are not desirable by other cavity-nesting species.

Eastern bluebird:

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- The development of successful translocation strategies that resulted in a bluebird population that initially grew following translocations; however, the population remained small and did not grow as expected leading us to describe the re-introduction as only partially successful.
- We suspect a combination of factors may be hindering population growth, including cavity availability and road mortality factors.
- The declining cavity and snag availability in years following Hurricane Andrew, perhaps in response to prescribed fires that consume more snags than they create.
- We failed to identify limiting factors in the population that could be addressed through habitat management.

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1980 - 2012: 32 years of re-introduction efforts of the hihi (stitchbird) in New Zealand

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Introduction

The hihi or stitchbird (*Notiomystis cincta*) is a rare New Zealand passerine listed as Vulnerable by the IUCN (2011) and as Nationally Endangered by New Zealand's Department of Conservation (Miskelly *et al.*, 2008). Hihi are the sole representatives of a New Zealand endemic bird family, the Notiomystidae that was historically widespread and common over the main North Island and surrounding offshore islands of the country. Following European colonization of New Zealand the hihi suffered a rapid decline in range and numbers until by about 1890 they had become restricted to a single remnant population on an isolated offshore island, Little Barrier or Hauturu (3,083 ha). The relatively unmodified forest ecosystem on Little Barrier supports a hihi population estimated to be between 600 to 6,000 birds. The rough terrain and isolation make reliable population estimates difficult although there are indications that hihi abundance has fluctuated on the island since the late 1800s. Beginning in 1980, an ongoing national recovery program aims to increase the range and numbers of hihi using re-introduction. Initially a captive population was established with the view it would provide most founders for re-introduction, but later harvesting from wild populations has provided most birds. To date there have been 21 translocations to eight

Male hihi © Eric Wilson

different locations (see: www.hihiconservation.com). Here we review this recovery program and highlight the mixed success but growing optimism surrounding management of this species.

Goals

- Goal 1: Identify sites favorable to the establishment of unmanaged or managed hihi populations and introduce hihi to the most favorable of these.
- Goal 2: Continuing to optimize management required to allow re-introduced hihi populations to persist in otherwise unsuitable habitats.
- Goal 3: Maintaining a captive population of hihi to produce birds for re-introduction.
- Goal 4: Coordinated and ongoing movement of hihi between populations as optimal management indicates (including, assessing suitable translocation sites, sustainable harvest for translocation, genetic management and reduced disease transmission risks).

Success Indicators

- Indicator 1: Coordinating a national approach to monitoring to ensure survival and breeding success is evaluated one year post release, and population growth is estimated from survival and reproduction data.
- Indicator 2: Essential that monitoring is designed such that results can be evaluated and management adjusted to ensure optimal hihi recovery (using an adaptive management model).
- Indicator 3: Existing translocated hihi populations are maintained and produce enough birds to be available as founders for new translocations.
- Indicator 4: That the captive population is a net producer of hihi for translocation and that hihi production costs are competitive until a more cost-effective or successful translocation technique is developed.
- Indicator 5: That global hihi numbers increase and are divided between multiple viable sites to protect against catastrophe in any one site.

Project Summary

Feasibility: Habitat requirements for hihi are difficult to identify and successful establishment at sites with populations of the more dominant bellbird (*Anthornis melanura*) and tui (*Prosthemadera novaeseelandiae*) remains the acid test for New Zealand restoration. However, the identification of key management initiatives that promote population persistence make re-introduction a feasible management approach. The New Zealand Government is committed to hihi conservation under the Wildlife Act and limited funds and extensive staff time from the Department of Conservation (DoC) are allocated to this project. Increasingly community based conservation groups are also becoming involved in hihi management as it shares the responsibility for management of this species and suitable sites. A national Hihi Recovery Group is convened by the DoC and includes DoC staff, community conservation group representatives and researchers. The purpose of the Hihi Recovery Group is to provide advice to hihi managers, including identifying and evaluating re-introduction sites. Currently the Hihi Recovery Group and managers of hihi populations benefits from generous

Little Barrier Island (Hauturu) © John Ewen

corporate sponsorship from Wesfarmers Industrial and Safety NZ Ltd. In all cases Maori iwi (tribal groups) are consulted and their support is obtained from both source and release sites prior to any translocation. Iwi have kaitiaki (traditional guardianship) over all native species and the locations where they are found in New Zealand.

Implementation: The practicalities and logistics of hihi translocations and subsequent monitoring have been refined over multiple translocation events. Early translocations were from the remnant population on Little Barrier whereas more recently hihi have been translocated from established re-introduced populations and the captive breeding facility. Husbandry techniques are well formalized and documented (e.g. Ewen *et al.*, 2011a) and disease risk assessments for translocation and preventative medications are continually revised (Ewen *et al.*, 2011b). The genetic ramifications of translocation have been assessed and recommendations have been made to best manage current genetic diversity and minimizing inbreeding accumulation (Brekke *et al.*, 2011). Another major transition has been from translocation to DoC managed reserves (mostly remote offshore islands) to community driven restoration projects (often mixed private and public lands on the main North Island). In all cases non-native mammalian predators are either controlled, or ideally, eradicated prior to hihi translocation.

Post-release monitoring: In most early re-introduction events translocated hihi were colour ringed but no standardized re-sighting protocol had been developed to accurately track their fates. Early monitoring inconsistently recorded persistence of release individuals, evidence of breeding of these birds and recruitment. Beginning in the early 1990s there was an effort to develop improved monitoring methods of individual survival, and in some cases individual reproductive success, across re-introduced hihi populations. This necessitated ongoing colour ringing of progeny in each re-introduced population and conducting regular and consistent re-sighting of ringed birds. These data are then used to model population growth and demographic responses to management (Armstrong *et al.*, 2002; Armstrong *et al.*, 2007; Chauvenet *et al.*, 2012). Current monitoring methods work well at accessible sites, however where release locations are remote or include difficult terrain the quality of monitoring data is considerably lower and remains a challenge. In addition, sites with mature forest that include natural nesting cavities for hihi make recording reproductive success more difficult than at sites where artificial nest boxes are used.

Major difficulties faced

- Lack of detailed knowledge of the resilience of the one remnant hihi population on Little Barrier to continual harvesting for translocation or the habitat features that allow the population to persist without supportive management.
- Difficulty in obtaining detailed post release survival data and ongoing survival and reproduction data from some populations. This is due to a mix of site characteristics (size and terrain), low density of hihi (at least initially), and monitoring skills of personnel.
- Poor population persistence without management at release sites and in at least one case uncertain population viability despite supportive management.
- Possible dispersal of hihi outside of protected areas at restoration sites located on the main North Island of New Zealand, and the difficulty in distinguishing the effect of dispersal against the impacts on the population of any predators that remain within an area despite control measures.
- Multiple problems in the captive breeding program including; (i) poor survival in captivity, (ii) low numbers of individuals can be housed in any single aviary due to aggression, (iii) high cost of maintaining the population and rearing young, and (iv) continual need for replacement of breeding birds from wild populations.

Major lessons learned

- Hihi can be easily caught, held and transported for re-introduction. Over time the techniques and husbandry requirements have been continually refined (details available in references or on request). A primary goal is to reduce stress to birds during all stages of translocation and to minimize the time taken for this process.
- Some hihi populations can grow with intensive supportive management. Currently all re-introduced hihi populations require some form of supportive management. Targeted monitoring is designed to evaluate and optimize management. This is important where management is costly and time consuming.
- Post-release monitoring is challenging but provides valuable information. The

Public release of hihi at Maungatautari 2011

Recovery Group is still developing a best approach to post-release monitoring for each site.
- Convening a national Recovery Group that includes DoC staff, community group representatives, iwi and NGOs, plus researchers benefits hihi conservation. The Recovery Group allows cohesive action with maximal input and agreement. It also provides an avenue to direct where research is required and also for generating funding.
- Direct translocations have proven to be a more successful and cost effective translocation technique compared to captive breeding, which has now been discontinued.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:
- Positive growth of re-introduced populations at some sites when supportive management provided a varying mix of; (i) provision of sugar water food supplementation, (ii) provision of artificial nest boxes at some sites, (iii) management of nest mite parasites and, (iv) ongoing control or exclusion of introduced mammalian predators.
- Despite supportive management some sites remain unsuitable, perhaps associated with disease (Mokoia Island population) or dispersal outside of protected areas (Ark in the Park project of Waitakere Ranges) or possibly from predation despite predator control attempts.
- All re-introduced populations require supportive management. The success of managed populations is due to the willingness of many determined groups and individuals to work together.
- The success of the project has been enhanced through collaborative information sharing and stakeholder involvement via the Hihi Recovery Group.
- Captive breeding program discontinued due to poor survival in captivity associated with disease and also an inability to hold many adult birds in the same enclosures due to aggression. Excessive cost to produce only few offspring relative to the ability to source large numbers of hihi for translocation from wild populations.

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Augmentation of the Puaiohi population through captive propagation and release on the Alakai Plateau, Kauai, Hawaii, USA

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Introduction

The Puaiohi, also known as the small Kauai thrush, is listed as Critically Endangered by IUCN/BirdLife and Endangered by the U.S. Fish and Wildlife Service. It is one of only two remaining Hawaiian species in the genus *Myadestes*, formerly comprising six Hawaiian species. (The Kamao, or large Kauai thrush (*Myadestes myadestinus*), has been considered extinct since 1989). The distribution of the Puaiohi (*Myadestes palmeri*) is currently limited to the upland, wet forests bordering the stream drainages of the Alakai Plateau, on the island of Kauai, Hawaii, at an altitude of approximately 1,050 - 1,500 m. Although endemic to Kauai and always uncommon, the Puaiohi is thought to have been more widespread and flexible in habitat use prior to the arrival of humans (Burney *et al*., 2001). Historical census data have been variable due to the lack of rigor in surveying effort. Based on limited surveys, the population in the mid-1990s was thought to "conservatively exceed 200 birds" (Snetsinger *et al*, 1999). Due to its diminished range and ongoing threats, a captive population was determined to be warranted as an assurance against extinction. This captive population subsequently

Puaiohi © Robby Kohley/SDZG

acted as the source for an augmentation program. The wild Puaiohi population is currently thought to number 300-600 birds (KFBRP, unpubl. data).

Overlooking the Alakai © Richard Switzer/SDZG

Goals

- Goal 1: Establishment of a captive population as an assurance colony to prevent extinction.
- Goal 2: Successful captive propagation, sufficient to provide cohorts for re-introduction.
- Goal 3: Soft release methods to ensure high levels of post-release survivability.
- Goal 4: Augmentation of the remnant Puaiohi population, through long-term survivability and successful reproduction.

Success Indicators

- Indicator 1: Successful development of effective techniques for the collection of wild eggs, artificial incubation, hand-rearing, captive maintenance and captive breeding of Puaiohi.
- Indicator 2: Captive propagation to provide 6 - 12 Puaiohi annually that are “surplus” to the goals of species assurance, constituting release cohorts.
- Indicator 3: Short-term survivability rate of ≥50% for 28 days post-release.
- Indicator 4: Follow-up survey data indicating an increase, or at least a halt in the decline, of the wild Puaiohi population; evidence of released birds showing long-term post-release survivability for multiple years; evidence of released birds breeding successfully in the wild.

Project Summary

Feasibility: In the mid-1990s, just as today, the remnant Puaiohi population was under pressure from a wide range of threats, including introduced predatory mammals (e.g. black rats, cats), avian-borne diseases carried by introduced *Culex* mosquitoes (avian malaria, avian pox), and the degradation of habitat by introduced ungulates (e.g. pigs, goats) and invasive plants (e.g. ginger). The majority of the Puaiohi’s remnant range is located in the Alakai Wilderness Preserve, on a plateau near Mount Waialeale. Although the wet, upland forest ecosystem of the Alakai is degraded by invasive plants and introduced mammals in some locations, it is still some of the best quality forest in Hawaii and in the Puaiohi’s historical range. Within the Alakai, native fruits (e.g. *Vaccinium*, *Cheirodendron*, *Styphelia*) and invertebrates - the two major food types in the wild

Puaiohi diet - are still found. Puaiohi nest primarily in niches on cliff faces running along river drainages. The territoriality of the species may limit the availability of nest-sites. Puaiohi are shy and secretive, presenting a challenge to survey and evaluate the impact of threats. The only total population estimate ever obtained for Puaiohi, from the early 1970s, is 177 $\pm$ 96 individuals (USFWS *et al.*, 1983). Research in the mid-late 1990s suggested that the population numbered more than 200 individuals, and the Puaiohi range possibly was expanding from an earlier contraction to only 20 km^2 (Snetsinger *et al.*, 1999). Further, more wide-spread, surveys indicated there were 300 - 500 Puaiohi in the wild by 2004 (Woodsworth *et al*., 2009). The altitudinal rise in the "mosquito line" - the elevation limit up to which the *Culex* mosquito and the malaria parasite (*Plasmodium relictum*) can exist - was considered an ever-increasing concern, likely putting further pressure on the Puaiohi's range.

Implementation: Due to the extinction threat to the Puaiohi, the initial goal was to establish a captive, assurance population at the Keauhou and Maui Bird Conservation Centers (KBCC and MBCC) - two captive breeding facilities operated by the Hawaii Endangered Bird Conservation Program, a partnership between The Peregrine Fund, the US Fish and Wildlife Service and the Hawaii Division of Forestry and Wildlife, which was subsequently to be operated by the San Diego Zoo Global. During 1996 - 1997, 19 Puaiohi eggs were collected from wild nests and transferred to a temporary incubation and rearing facility on Kauai. Fifteen eggs were viable and resulted in the hatching of 15 chicks. When robust enough to travel to the Big Island, the chicks were transferred to aviaries at KBCC, as the founders for the captive breeding program (Kuehler *et al.*, 2000). Between 1998 and 2011, the captive breeding program produced a further 420 viable/fertile eggs and hatched 336 more chicks, of which 268 were raised to independence.

Between 1999 and 2012, 225 Puaiohi were transferred to either the Kawaikoi, Koaie or Halepaakai river drainages within the Alakai, constituting 14 release efforts. In preparation for leaving MBCC or KBCC, each bird was given a full veterinary exam to ensure they were fit for release and carrying no pathogens. Birds were transported to Kauai by inter-island plane and then transferred to the remote Koaie or Haleapaakai release sites by helicopter, or by vehicle and on foot to the more accessible site at Kawaikoi. Birds were installed in 2.4 m x 2.4 m x 2.4 m pre-release aviaries, elevated on a predator-proof scaffold approximately 1.5 m above the ground. The area immediately surrounding the aviaries was baited with rodenticide. Up to 6 birds were held in each aviary for a pre-release acclimation period of 7 - 15 days. Native berries and vegetation were provided in abundance. Several days prior to release, each bird was captured, examined, weighed, banded, and whenever appropriate, fitted with a radio-transmitter. Any birds presenting concerns at this point were not released and three birds died in the aviaries before release.

Of the 222 birds released, 176 (79.1%) were released at under one year of age. The mean age at release was 445 days. Each cohort was released by simply opening the hatch, with birds leaving the aviary at their own pace. Supplemental food was provided in and around the open release aviaries for up to 1 month, to facilitate the birds' transition to wild foraging.

Puaiohi chicks © Sharon Belcher/SDZG

Post-release monitoring: Whenever possible, birds were monitored using radio-telemetry up to 28 days post-release. The monitoring effort varied according to adaptive management of release methods and availability of personnel. In some years, helicopter surveys aimed to detect birds which had dispersed long distances. During the initial 1999 release at Kawaikoi, all 14 birds were monitored to assess survival, dispersal, and home-range establishment - all 14 birds survived up to 56 days post-release (Tweed *et al.*, 2003). In later years, the percentage of release birds with confirmed status at 28 days was as low as 8.3%. Of 222 birds released in total, 122 (55.0%) had confirmed status at 28 days post-release. Of those 122 birds, 80 (65.6%) were recorded as alive. General population surveys have resulted in longer-term observations of 20 released birds - paired up, nesting and raising at least 24 chicks to fledging (Tweed *et al.*, 2006; KFBRP, unpubl. data).

Major difficulties faced

- Although managed by the State of Hawaii as a Wilderness Preserve, the Alakai is operated for multiple purposes, including recreation and hunting. Based on the scarcity of funding and lack of feasibility for ecosystem restoration, minimal effort was made to restore Puaiohi habitat to its pristine state. It is acknowledged that this represented both a less-than-perfect scenario and a challenge in the re-introduction effort. Additionally, throughout the course of this re-introduction program, it appeared that mosquitoes had invaded further into the Puaiohi's remnant range.
- The largest sample group of released birds consisted of those with unknown status at 28 days, represented by 100 birds (45.0%). The thickly forested ridges, peaks and drainages of the Alakai plateau made radio-telemetry a challenge and impacted the evaluation of post-release survival and dispersal.
- There was low detectability of released birds long-term beyond the period of the radio-telemetry effort. Some data exists on the detectability of banded, wild birds, but this only reflects the pairs living at higher densities within the core distribution. This limitation makes it challenging to evaluate the long-term benefit of the released birds on the wild population.
- At the start of the re-introduction effort, little precise data existed on the reasons for range retraction, population and density of the Puaiohi. It is

possible (but by no means certain), that core Puaiohi habitat was at its holding capacity for territories, while habitat at the periphery of the range was being encroached upon by a number of threats. Therefore re-introduction in conjunction with only targeted amelioration of the threats impacting recruitment may have had little benefit.

- By the end of the intensive captive breeding program in 2011, several chicks showed signs of congenital abnormalities, presumed to be related to inbreeding depression. This should not be a surprise, based on more than 5 generations of reproduction descended from only 14 wild founders and their 15 eggs, resulting in a decline in the genetic diversity of the captive flock to only 84%.

Major lessons learned

- The harvest of wild eggs, artificial propagation and productive captive breeding of the Puaiohi provided a valuable source of potential new recruits for the wild population.
- Although the initial captive breeding program was established as an emergency measure to prevent extinction, the re-introduction effort would have benefitted from higher quality and more rigorous data on population, range, ecology, reproductive success, limiting factors, epidemiology of introduced disease, and other threats. This would have provided greater guidance for the re-introduction effort.
- Attempts to augment sinking populations or establish new populations are much more effective if conducted in combination with other measures to ameliorate the limiting factors.
- Similarly, the re-introduction program would have been more effective if other measures had been taken within a holistic approach to ecosystem restoration.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- The establishment of the captive assurance colony was highly successful (particularly for a captive passerine program), with 222 birds provided for release.
- Near the midpoint of the reintroduction program, data collected from 2003 - 2006 indicated that there were 300 - 600 Puaiohi in the wild, and continuing research suggests that the population is probably stable (KFBRP, unpubl. data). Even if the population estimates are imprecise due to difficulties in acquiring census data, it can be concluded that, at best, there was an increase in the wild population, and at worst, there was no significant decline in the population. This is within the context of other endemic species in the Alakai declining drastically over the same time-frame.
- Although it is challenging to assess the long-term contribution of released birds to recruitment and population increase, at least 20 released birds successfully bred in the wild.

- The re-introduction program provided the stimulus and momentum for more extensive research of wild Puaiohi and other endemic species in the Alakai, which is currently ongoing.
- Failure to tackle the major threats that are assumed to impact the wild population has resulted in no significant increase in the range of the Puaiohi.

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Field propagation and release of migratory Eastern loggerhead shrike to supplement wild populations in Ontario, Canada

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Introduction

The loggerhead shrike is one of two species of shrikes (Laniidae) in North America, and the only shrike with an exclusively North American range. The species has undergone one of the most persistent and drastic population decline of any North American passerine and is now largely extirpated from northeastern North America, with the exception of a very small population in Ontario, where recovery actions are focused. The Ontario population is classified as the subspecies *Lanius ludovicianus migrans*, however recent research suggests it represents a unique genetic cluster significantly different from *L. l. migrans* (Chabot, 2010). The known number of breeding pairs in the province ranged from 20 to 35 pairs in the past decade and the global population is likely less than 100 individuals. The loggerhead shrike is protected internationally (Canada, Mexico and USA) by the Migratory Birds Convention Act (1916), and *L. l. migrans* is listed federally in Canada as Endangered under the Species at Risk Act. Globally, the species is classified as Apparently Secure (IUCN). This passerine is unique in its predatory and impaling behaviors, and is equipped with a raptor-like beak. It is often

Loggerhead shrike © Larry Kirtley

considered a ‘flagship’ species for grassland birds, a guild of high conservation concern.

Shrike Chicks (10 days old) © Tracy Anderson

Goals

- Goal 1: Preserve the genetic diversity of loggerhead shrike in eastern Canada.
- Goal 2: Provide a suitable source of birds for re-introduction to augment the wild population.
- Goal 3: Undertake research to increase the effectiveness of the breeding and release program (apply adaptive management).
- Goal 4: Undertake other research to enhance management of the wild population, e.g. disease (West Nile virus), diet, toxin studies, dispersal and migration studies (radio-tracking, geolocators).
- Goal 5: Re-establish a viable, self-sustained, and broadly distributed wild population of *L.l. migrans* in the current population range and re-establish the species in parts of its historical range in Canada.

Success Indicators

- Indicator 1: Breeding of *L.l. migrans* in captivity.
- Indicator 2: Survival and return of released captive-reared birds to breeding grounds in Canada.
- Indicator 3: Captive-reared birds breeding in the wild.
- Indicator 4: Captive population retains genetic diversity (maintain 90% of the genetic diversity of the founder population for 25 years).
- Indicator 5: An increase in the number of breeding pairs in the wild in Canada.

Project Summary

Feasibility: Population monitoring began in the 1980s with annual comprehensive surveys starting in the early 1990s. Evaluation of habitat availability indicated the presence of extensive suitable but unoccupied habitat in the historical range in Ontario. At the time this program was established, captive breeding of shrikes was already underway for other subspecies: *L. l. mearnsi* and *L. l. excubitorides*. Release site selection considered habitat suitability at the site, extent and proximity of other suitable habitat, projections of land-use over time, logistics, community support, and the existence of wild pairs in the region.

Implementation: Founder stock was obtained as nestlings (n = 48) from wild pairs in 1997 and 1998. Breeding occurred at the Toronto Zoo (Toronto, Ontario) and McGill University (Ste Anne de Bellevue, Quebec) initially to obtain a viable captive population (i.e. maintain genetic goals). In 2001 an experimental *in-situ* field propagation program was initiated. Current release efforts focus on supplementing existing populations rather than re-establishing locally extirpated ones. Pairs are bred in large field enclosures within suitable shrike habitat; while food (crickets, mealworms and hopper mice) is provided, birds are also able to hunt wild prey entering through the mesh walls, and are exposed to natural predators (e.g. merlin (*Falco columbarius*)). Young are soft-released into their natal territory and receive supplemental food. Young are also produced at *ex-situ* breeding facilities, with young "hacked out" at the field sites. From 2001 - 2012, 663 juvenile shrikes were released with ca. 100 young released annually in some years (2006 to 2010). Thirty confirmed sightings of returning birds have been made to date, with the first captive-reared bird returning to successfully breed with a wild shrike in 2005 (Nichols *et al.*, 2010). The return rate observed for captive-reared shrikes is in-line with that reported for wild juveniles in Ontario and elsewhere, and productivity in the wild is similar to that of wild pairs in the province (Lagios *et al.*, in press). Evaluation of release techniques (e.g. release group size, age) has allowed for adaptive management to improve results in terms of returning birds (Lagios *et al.*, submitted).

West Nile virus (WNV) and *Capillaria sp.* (nematode parasite) have been the major identified disease concerns for the program. The captive population is vaccinated annually against WNV since 2008. *Capillaria* exists normally within the wild population; routine fecal/pellet screening and treatment aim to reduce parasite burdens in captive birds rather than completely eliminate it. From 2007 - 2011 the program experienced substantial fledgling mortality and deaths of young (<4 years old); necropsy results were largely inconclusive but ruled out an infectious disease agent. In 2010 we began consultations with epidemiologists to further investigate causes but no obvious cause was identified. The issue is likely multifactorial including environmental conditions, stress from double clutching and cage density. Collaboration with epidemiologists to identify and mitigate contributing factors continues.

The captive breeding and release program is tightly integrated with other recovery activities, e.g. wild population monitoring, color banding, habitat restoration and stewardship, outreach and education, and research to address knowledge gaps. An external review of the program (Kleiman & Lynch, 2008) concluded that the release program had achieved success relatively early (as defined by returning birds successfully breeding in the wild) and suggested that the program would ultimately provide a model for future recovery programs for other shrike populations in North America and other at risk migratory passerines.

Post-release monitoring: All released birds are colour banded; since 2009 a unique 4-colour combination has been used. Immediate post-release monitoring follows birds as they disperse from the field site and intensive field staff surveys spot returning birds in subsequent seasons. These efforts are supplemented by a volunteer Adopt-a-Site program and additional outreach to the birding community to "Spot a Shrike, Save a Species". A network of partners and volunteers was established in the U.S. to aid in detection of banded shrike along migration routes and on wintering grounds. There have been several recent sightings of banded birds in the U.S., two of these captive-reared birds appearing in Ohio and northern Virginia during migration.

Observing captive loggerhead pairs
© Wildlife Preservation Canada

A radiotelemetry study found that shrikes tolerate radio-transmitters (and behave normally) and we observed 75% survival of release young pre-migration (Imlay *et al.*, 2010). Birds moved too quickly out of Canada to track migration routes, even with the use of aerial telemetry. To locate the wintering grounds, geolocators were deployed on 108 shrikes from 2009 - 2011. Although 3 geolocator birds returned to Ontario and were successfully re-trapped, in all cases the devices had failed prior to onset of migration. Further trials with captive shrike suggest that the failures are not caused by the shrikes themselves and we will continue this study in 2013.

Major difficulties faced

- Migration routes and wintering grounds for the Ontario population are unknown, therefore we cannot yet monitor birds or determine threats outside of the breeding season.
- Unexplained mortality in the captive population undermines its genetic viability and the numbers released.
- It is difficult to maintain community goodwill towards the release program when there is a lack of clarity surrounding enforcement of provincial endangered species legislation and the federal Recovery Strategy which identifies Critical Habitat on private lands; community polarization impacts property access and monitoring efforts.
- Sustained funding to maintain a full-scale and long term program, that crosses geo-political boundaries.
- Issues with field surveys include high dispersal of young, nesting on private properties, low detectability of shrikes.

Major lessons learned

- Successful breeding and high productivity achieved through *in-situ* breeding conditions.
- Young that are parent-reared *in-situ*, and provided with live prey, demonstrate a full range of natural behaviors and show a high rate of survival post-release.

- Due to the species' foraging behavior, cage height was an important dimension in reducing stress and increasing breeding success and productivity (increasing cage height from 2.4 m to 3.0 m had positive results).
- Ongoing communication, good-will, and partnership development with stakeholders is key; this includes local landowners, and industry, e.g. beef, aggregate, renewable energy (wind and solar).
- Successful recruitment of captive-reared young into the wild population over successive years shows captive breeding is a viable conservation tool for migratory birds.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- Production and release of large numbers of young annually.
- *In-situ* breeding conditions, soft-release and supplemental food.
- Integration of release program with other recovery activities, including surveys of wild population, stakeholder engagement and habitat stewardship.
- Recruitment of captive-reared individuals depends on their successful migration, wintering in an unknown location, and return to breeding grounds.
- Despite the successful captive breeding program; the wild population has not increased in size due to threats during the non-breeding season.

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Re-introduction of the oriental white stork for coexistence with humans in Japan

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Introduction

The endangered oriental white stork (*Ciconia boyciana*) (OWS) is distributed within the Far East with a global population size estimated to be between 1,000 to 2,499 birds (IUCN, 2012). In Japan the OWS was common up until the first half of 19th century, but declined in number thereafter due to human persecution. The last wild population persisted for a time in the Tajima District, in the northern part of Hyogo Prefecture, but had died out completely by 1971 due to widespread pesticide application. The Government of Hyogo Prefecture had established a captive population in the 1960s, and successful breeding started in 1989. With an increasing captive population size, the government planned a re-introduction project aiming at restoring the harmonious coexistence between humans and storks. All these programs concerning conservation and re-introduction of the OWS have been practiced under close cooperation with the Agency for Cultural Affairs, Japan. The first releases of captive-bred bird by the Hyogo Park of the Oriental White Stork (HPOWS, established 1999 by Hyogo Prefecture) took place in Tajima in the autumn of 2005 (Ohsako *et al.*, 2008).

Goals

- Goal 1: Re-establishment of the Japanese OWS population in coexistence with humans.
- Goal 2: Establishment of a meta-population structure in Japan.
- Goal 3: Linkage with the continental populations to fuse them into a meta-population.
- Goal 4: Contribution to the global conservation of the species.

Oriental white stork

Stork habitat - co-existing with humans

Success Indicators

- Indicator 1: Survival of released birds.
- Indicator 2: Reproduction in the wild.
- Indicator 3: Establishment of other local populations in Japan.
- Indicator 4: Maintenance of genetic diversity within the birds in the wild.
- Indicator 5: Understanding and cooperation by local communities promoting coexistence with the storks.

Project Summary

Feasibility: The original habitat of the OWS is floodplain where intensive rice cultivation has been taking place for a long time in Japan. Wild storks in the past naturally foraged within paddy fields and they were regarded as a nuisance through their trampling of seedlings of the rice plant. Thus it was a key challenge to persuade the local people to support the re-introduction. The project was based on the IUCN Guidelines for Re-introduction (IUCN, 1998) and developed into an action plan in 2003 under the slogan “Environment where storks can live is also safe and secure for humans” (CROWS, 2003). With the agreement of the local community a Liaison Committee for Re-introduction was organized with participation from all stakeholders, and efforts were made to improve both the natural and the social environment, e.g. restoration, education, and a newly developed cultivation method that helps production of prey animals.

Implementation: Twenty-seven storks were released by HPOWS between 2005 and 2010 within the rural area of Toyo-oka City, situated at the northernmost part of Tajima District. Artificial nest towers last used by wild birds in the 1960s, were renovated by stakeholders with the result that the wild storks again use them for nesting. Some birds were artificially fed to encourage their settlement, and a significant number of birds foraged in an open cage within the property of HPOWS where prey fish are supplied every day to flightless display storks.

Post-release monitoring: Monitoring and scientific studies have been conducted by HPOWS researchers. Of the 24 released birds (3 of the 27 were taken back into captivity again) 16 survived at the start of the 2012 breeding season as the first generation in the wild, a 67% overall survival rate. The first pair was formed in 2006 and the number of pairs increased from 2 in 2007, 5 in 2008, 6 in 2009, 7 in 2010 and 9 in 2012. The first fledgling of the second generation was thus produced in 2007 and by 2011 a total of 36 second generation birds had fledged, with 29 of them still alive at the start of the 2012 breeding season. The annual survival rate of young after fledging is as high as 81%. With the addition of a female believed to have immigrated from the continent, the total population size has increased to 46. By monitoring an almost fully banded population, it was shown that this species has territories defended by pairs throughout the year, and immature birds younger than four years live as floaters. Based on scientific analysis of monitoring data a grand-design for re-introduction was developed by HPOWS (2011).

Although young birds fly long distances and visit various districts of Japan, they usually return to Tajima centered by Toyo-oka Basin after a short stay in each district, possibly indicating some difficulty in natal dispersal and suggesting low food availability in rural areas where bio-productivity has declined due to a change in the water-supply system for paddy fields all over Japan. Other than this, the limited genetic variability of released birds, with highly biased breeding success among pairs, increases the probability of inbreeding in the wild. Genetic analysis of skins of the past wild birds (Murata *et al*., 2004) suggests that they were in the midst of an extinction vortex due to inbreeding (HPOWS, 2011). In order to lower the probability of inbreeding in the present population, birds belonging to new families were added to the captive population, with some of them being released in 2012. In addition, various attempts are conducted to reduce the birds' use of artificial feeding (Ohsako & Ezaki, 2011).

Another problem is the fact that some pairs use nest-towers built at the center of a paddy field, completely in open space. As the past wild population nested in pine trees on hillsides, pairs were invisible to each other as long as they stayed on the nest. But with introduction of nest-towers just before the extinction of the wild population, birds started to nest in open space and this arrangement is considered by local people to be the normal situation. After re-introduction, eggs and chicks in tower nests are sometimes attacked by neighboring pairs while the parents are absent from the nest, indicating that those nest-towers should be moved to hillside. In order to solve this problem HPOWS experimentally moved a nest-tower just before 2012 breeding season. Efforts to establish other local populations also have started. A pair of captive storks were lent from HPOWS to Fukui Prefecture where the local government is now engaged in captive breeding with the aim of supporting re-introduction in the future. Lastly, in


Third generation wild storks

2012, a total of four birds of the third generation fledged from two nests, one of which was reared completely free from artificial feeding. We call this the "first genuinely wild" bird.

Major difficulties faced

- Poor prey animal communities, especially that of fish.
- Dependence of storks on artificial food.
- High probability of inbreeding.

Major lessons learned

- It is very difficult to restore the Japanese rural environment in the face of bio-productivity declines due to a changing system of paddy field irrigation. But it must be restored at least partially using any possible engineering methods, as this will contribute not only to re-introduction but also to conservation of local biodiversity, especially in aquatic animals.
- It is easy for the storks to depend on artificial feeding and it is difficult to reduce reliance on that, mainly because of social reasons, especially the mind of local people loving the beautiful bird.
- Education is important to persuade people that wild animals are not free from death, especially when they are young.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- Acclimatization and training were successful due to the high adaptability of the storks.
- It is easy for the storks to form pairs in the wild, whereas it is difficult to achieve this in captivity.
- The storks are attractive and iconic enough to promote regional development and local environmental action.
- Establishment of the Liaison Committee for Re-introduction by Hyogo Prefecture.

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Re-introduction of red-billed oxpecker in central Zimbabwe

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Introduction

The red-billed oxpecker (*Buphagus erythrorhynchus*) ranges from southern Africa to north-eastern Africa. Although its conservation status is Least Concern (IUCN Red List), its population range and number are declining. Population decreases followed declines in the populations of the hosts (e.g. buffalo, black rhinoceros) on which the oxpecker perches to feed, eating ticks; and the widespread dipping of cattle in acaricides (based on arsenic, organochlorine, or organophosphate compounds) that kill both ticks and oxpeckers. Recently, dips that control ticks but which do not kill oxpeckers (Amitraz ,or pyrethroid-based) have become available (Couto, 1994). The red-billed oxpecker was re-introduced to Shangani Ranch to aid in tick control on both cattle and wild animals. Shangani Ranch (*c*. 480 km^2) lies on the highveld in central Zimbabwe. Mean annual rainfall is *c*. 600 mm, with a single rainy season during November - March (Dunham *et al*., 2003). The main vegetation types are: *Terminalia sericea* woodland and wooded grassland; *Brachystegia–Julbernardia* woodland; *Colophospermum mopane* woodland and shrubland; *Acacia-Combretum* woodland on alluvial soils; and hydromorphic grassland. The ranch is used primarily for beef production, but supports significant populations of wild herbivores, including impala, kudu, tsessebe, eland, zebra and giraffe, as well as predators such as leopard, cheetah and jackal.

Captured oxpecker being measured after removal from mist net © Tracey Couto

Goals

- Goal 1: Create a self-sustaining population of the red-billed oxpecker on Shangani Ranch.
- Goal 2: Establish the red-billed oxpecker as a natural means of tick

control on Shangani Ranch, so that the use of chemical acaricides could be reduced or eliminated.

Cattle a major host for released oxpeckers on Shangani Ranch © Clive Swanepoel

Success Indicators

- Indicator 1: Sightings of oxpeckers on the ranch during the days and weeks after their release, indicating that the birds were alive and that they had not left the release area.
- Indicator 2: Sightings of juveniles in the years following the release, indicating that oxpeckers were breeding on the ranch.
- Indicator 3: Increased frequency of sightings across the ranch during the years after the release, indicating that both the population number and range were increasing.
- Indicator 4: A decline in the number of ticks on the ranch's cattle, suggesting that oxpeckers were controlling tick numbers.

Project Summary

Feasibility: The re-introduction was undertaken jointly with the Wildlife Unit of the Veterinary Research Laboratory and the Ornithology Unit of the then Department of National Parks & Wild Life Management, which determined the criteria for oxpecker re-introductions (Mundy, 1992). These were that: no purple-label (very toxic to mammals) acaricidal dips be used within 10 km of the release site; the release area contain >500 tick-infested large herbivores (i.e. wild herbivores, or undipped cattle) in >30 km^2; and at least 20 birds are released (recognising that birds roost communally, and that 'helpers', probably youngsters from an earlier brood, assist a breeding pair at the nest). Use of purple-label dips on Shangani Ranch had been discontinued, the ranch was big (480 km^2) and large, wild herbivores were numerous (>7000 - mainly impala, tsessebe, warthog and kudu - were counted during a helicopter-based, total-area survey during 1993).

Implementation: Twenty-two red-billed oxpeckers (19 adults and 3 sub-adults) were captured at Rukometjie Research Station in Mana Pools National Park, northern Zimbabwe, during 30th September - 1st October 1992. This was outside the breeding season, which coincides with the rains. The birds were caught during the early morning in mist nets erected near a pen containing a small herd of cattle that was used during tsetse fly research (Couto, 1994). When caught in the net, an oxpecker would often utter distress calls that attracted other members of the family group, which then became caught. After removal from the net, each

Captured oxpeckers in temporary aviary prior to transfer to release area © Janine Walls

bird was weighed, measured, ringed, checked for moult, and its age determined. While the oxpeckers were handled, their feet were wrapped in small pieces of adhesive tape to protect the researcher from the needle-like claws and thus make handling easier.

The captured oxpeckers were placed in a collapsible aviary measuring 2.4 m x 2.4 m x 1.2 m and sited in a secluded, shady position. The aviary was constructed with a tubular-steel frame and 1 cm wire-mesh sides. A small doorway in one panel provided access. Shade cloth covered part of the aviary to provide extra shade and reduce external stimuli. Perches were provided in the corners and drinking and bathing water was freely available in shallow troughs on the ground. In the centre of the aviary, a 200-litre metal drum was placed on its side, with sack cloth tied around it to simulate the body of an animal. On the top of the drum were secured plastic Petri dishes that were filled three times daily with unclotted bovine blood. The birds were also fed a mixture of chopped liver, egg and Pronutro (a commercial meal for people, manufactured from maize, soya and sugar, with added yeast, minerals and vitamins), spoonfuls of which were placed on the drum.

Catching the oxpeckers in the aviary could be stressful for the birds. First, the perches, drum and other obstructions were removed and then the birds were compressed into one end of the aviary with a sheet attached to an aluminium frame just less than the width of the aviary. Once caught, the birds were placed in small, portable cages for transfer by road to Shangani on 2nd October. There, the birds were placed in another aviary, where the food was similar to that at the capture site. Because the birds arrived at Shangani after dark, they were left in their travel boxes overnight and these were opened at first light the next morning. Two birds had died during transit or overnight, and a third died soon after the birds were freed into the aviary. The remaining oxpeckers were caged for four days and fed and watered daily. On 6th October, nineteen birds were caught in the aviary, transported in boxes to the centre of the ranch (a 40-minute journey by road) and released. They were freed close to a herd of cattle, on which they soon settled, prompting the cattle - ignorant of oxpeckers - to run off.

Post-release monitoring: Post-release monitoring of oxpeckers involved incidental observations of birds by staff during cattle management operations, e.g.

when cattle were rounded up. Sightings provided information about the spatial distribution and dispersal of released birds. For example, one bird was seen 17 km from the release site about three months after release. And during the first six months of 1995 (27 - 33 months after the release) ranch staff recorded groups of 1 - 9 oxpeckers on 52 days. The first sign of breeding was a nest in an *Acacia* tree during November 1993, and later the presence of juveniles in recorded groups indicated continued successful breeding. Oxpeckers were often on cattle, but also on kudu and warthog. Birds were observed on neighbouring ranches and during 1998 five were seen approximately 90 km from the release site (Couto *et al.*, 2000).

Major difficulties faced

- Some neighbouring cattle ranches continued to use purple-label (high mammalian toxicity) acaricides.

Major lessons learned

- The oxpeckers were transferred by road to Shangani and arrived there in the evening, after dark. They were left in their travelling boxes overnight and released into the aviary at first light the next morning. Two birds had died during transit, or overnight. The other birds flew straight to the water, where a third died. Subsequent observations revealed that the birds did not feed during the early morning, or late afternoon - hence they could not have been fed or watered immediately after they arrived at Shangani, because by then it was dark. The road journey had been long and these deaths emphasized the importance of oxpeckers being regularly fed and watered and checked for heat stress while in transit.
- The oxpeckers in the aviary at Shangani spent much time on a high ledge and this observation suggested that a good way to catch the caged birds was to position roosting boxes with closeable entrance holes high in the aviary. The roosting birds could be shut in the boxes during the night prior to their transfer and, the next morning, the roosting boxes could be used as travelling boxes to move the birds to the release site.
- The re-introduced population of red-billed oxpeckers had no major impact on the numbers of ticks found on cattle, although the fluctuations in tick numbers, especially of the brown ear tick, seemed to be less extreme and less frequent than previously. Recent studies have shown that tick loads on cattle are unaffected by oxpeckers and that oxpeckers significantly prolong the healing time of wounds by feeding on blood at the site of these wounds (Weeks, 2000). Hence, regular dipping of cattle with green-label (low mammalian toxicity) acaricides continued to be necessary at Shangani. The presence of oxpeckers dictates the need to avoid resuming the use of purple-label chemicals and instead anti-helminthic injections were given to cattle.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- The social behavior of oxpeckers (communal roosting, helpers at the nest) prompted the decision that a release group contain at least 20 birds (although after three deaths, 19 were freed at Shangani). All the birds were captured at the same site during a short period, in the hope that the birds were from the same social group(s).
- The release site was near the centre of a large ranch that contained numerous wild ungulates, as well as cattle that were dipped with green-label (low toxicity) acaricides. Hence the release ranch was big enough to sustain an oxpecker population, even though some birds may have died after moving to neighbouring ranches, some of which were still using purple-label acaricides. Nonetheless, the primary goal - of establishing a self-sustaining population in an area from which the species had disappeared - was achieved.

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Brown treecreeper re-introduction into eucalypt woodland in the Australian Capital Territory, Australia

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Introduction

The brown treecreeper (*Climacteris picumnus*) is a small ground-foraging bird endemic to eastern Australia. The species is listed as vulnerable under the Australian Capital Territory *Nature Conservation Act 1980* and the New South Wales Threatened Species Conservation Act 1995 (subspecies *victoriae*). The brown treecreeper is a facultative cooperative breeder, living predominantly in gregarious social groups comprised of a breeding pair and a number of offspring that have delayed dispersal. The species nests and roosts in tree cavities and is almost entirely insectivorous. There is evidence of dramatic declines of this species throughout its range. The main causes of decline are considered to be habitat

Brown treecreeper © Veronica Doerr

degradation, such as the loss of tree hollows and components of high quality ground-foraging habitat such as coarse woody debris and ground litter. Further, habitat fragmentation significantly disrupts the recruitment of females owing to the short-distance dispersal capabilities of the species. Female offspring tend to disperse earlier and further than males, however this is generally only a distance of 1 - 2 km. The brown treecreeper was re-introduced into Mulligans Flat and Goorooyarroo Nature Reserves, which are two connected eucalypt woodland reserves that are undergoing ecosystem restoration in south-east Australia. The species was recently locally extinct from these reserves.

Goals

- Goal 1: Installation of experimental restoration treatments in the nature reserves where the re-introduction was to be conducted. The re-introduction and subsequent survival, behavior and habitat use of re-introduced individuals was used to assess the success of these restoration treatments.
- Goal 2: Successful selection and translocation of brown treecreeper social groups from the source population to the re-introduction site.
- Goal 3: Establish a self-sustaining population based upon survival and reproduction indicators.
- Goal 4: Intensive monitoring of re-introduced individuals to obtain information on brown treecreeper behavior, movement and habitat use.
- Goal 5: Examination of the factors influencing the outcome of the re-introduction.

Success Indicators

- Indicator 1: Successful release of brown treecreepers.
- Indicator 2: Survival rate over time: survival of 70% of re-introduced adult birds 3 days after release, 50% at 4 weeks after release, and 40% at 1 year after release.
- Indicator 3: Successful reproduction, with the survival of at least one young to fledgling from at least one social group within two years.
- Indicator 4: Detailed examination of data collected on brown treecreeper survival, behavior, movement, and habitat use to provide unique information regarding this species in an unfamiliar, experimentally restored environment.

Project Summary

Feasibility: Brown treecreepers were sourced from populations 200 km west of the release site, in the Murrumbidgee region of New South Wales, Australia. These source populations were the most appropriate for re-introduction as they had been studied extensively, the birds' social relationships were known, and the populations were considered sufficiently stable and large to allow the removal of some individuals without compromising the stability of the populations.

The reserves where re-introduction took place, Mulligans Flat and Goorooyarroo Nature Reserves (measuring 1,623 ha), are the site of a large long-term restoration project. Habitat manipulations in these reserves included the addition of 2,000 tons of coarse woody debris, management of variation in ground vegetation cover through kangaroo exclusion areas, and the installation of brown treecreeper nest boxes. These manipulations were considered to be beneficial in reversing habitat degradation, which was the likely cause of local decline for the brown treecreeper. The restoration manipulations were implemented as experimental treatments to enable examination of the effects of treatment combinations on brown treecreeper survival, movement, behavior and habitat selection.

Mulligans Flat Nature Reserve settlement site © Victoria Bennett

Implementation: Brown treecreepers were re-introduced in November 2009. Birds were released in social groups containing dependent fledglings to maintain bonds between individuals in a group. One social group was captured and released per day as a hard release to minimize handling time and avoid unnecessary stress. Seven brown treecreeper social groups were re-introduced, comprised of 43 individuals (26 adults and 17 fledglings). Each individual was fitted with a unique combination of coloured leg-bands. Additionally, radio-transmitters were fitted to the breeding female and one or two helper birds per social group (total of 18 adult birds). The re-introduction was performed experimentally, by releasing different social groups in areas that had been subject to different habitat restoration treatments. Social groups were released in areas with combinations of two experimental treatments: i) high or medium level of ground vegetation cover, and ii) the presence or absence of nest boxes.

Post-release monitoring: Released birds were monitored daily until February 2010, with observations recorded for survival, location, behavior and substrate use. Ongoing visual monitoring of survival was conducted until March 2011. Short-term post-release survival rates were high over 24 hours (93%) and 3 days (91%), with high levels of social group cohesion maintained. The number of adults and fledglings confirmed alive over the initial three month monitoring period steadily declined. There were no apparent differences in survival between males and females, adults and fledglings, social groups, or between birds that carried radio-transmitters and those that did not. The coarse woody debris supplemented to the reserves appeared to benefit the brown treecreeper through influencing their behavior and significantly increasing the probability that an individual would forage on a log or on the ground. This potentially occurred through influencing the individual's foraging efficiency. However, variation in ground vegetation cover did not influence the species' behavior and substrate use. This may have been caused by the overall limited use of the ground layer, particularly in relation to previous studies on the species. This suggests that the degrading processes previously acting in the reserves have not been sufficiently reversed.

All radio-tracked brown treecreepers were recorded leaving their immediate release site, irrespective of the habitat experimental treatments at their release sites. This result suggests that re-introduced individuals may always explore their surroundings regardless of the quality of the habitat they are provided with. Individuals were observed moving extensive distances, with significant variation in search area among individuals. This may be a result of the re-introduction process, or indicate a rejection of the release site. However, the result also indicates that re-introduced individuals are likely to be able to adjust their movement behavior and find suitable habitat. Settlement of social groups was significantly affected by the level of ground vegetation cover, with dry forest areas with low vegetation cover having the highest proportional rate of settlement.

Despite the experimental restoration conducted within the reserves and attempts to conduct the re-introduction within a best-practice framework, the re-introduction failed to meet all of the predetermined criteria for success. This was particularly the case for medium-term survival. Further examinations were conducted to compare the habitat within the nature reserves where the re-introduction took place and the habitat at the source population. Although predation appeared to play a key role in bird survivorship, there was no significant difference in predation pressure identified between the two habitats. However, re-introduced individuals may have been particularly vulnerable to predation because of an increased flight time to reach a refuge area when under threat due to a lower number of refuge areas in the re-introduction reserves compared with the source sites. A lower ground foraging habitat quality was also identified at the release sites, however, brown treecreepers were able to disperse extensively throughout the reserves and settle in areas with generally higher-quality foraging habitat.

Major difficulties faced

- Much lower survival rates of re-introduced brown treecreepers over the first year than have been reported in any naturally occurring population of brown treecreepers.
- Higher predation levels of released individuals than expected, particularly by native avian predators. All known deaths of radio-tracked birds (n = 4) appeared to be due to predation by native predators. To a smaller extent, elevated densities of aggressive species such as the noisy miner (*Manorina melanotis*), negatively influenced brown treecreeper releases and may act as a barrier to re-colonisation by other locally extinct species.
- Released individuals were observed dispersing across extensive distances that were greater than distances previously observed among brown


Social group just before release © Peter Mills

treecreeper natal dispersers. This was despite existing studies examining brown treecreeper natal dispersal in detail. Extensive dispersal caused logistical difficulties in effectively monitoring all individuals and confirming their survival, despite radio-tracking, and particularly once radio-transmitters were no longer functioning and during and after the time when the species normally disperses. Hence, the disappearances of some individuals could be a result of dispersal, not just death.
- The re-introduction was a large logistical project requiring extensive organization of licensing, acquisition of funds, equipment and personnel. This project was conducted as a post-graduate project and was therefore somewhat restricted in the amount of monitoring conducted. Therefore, there were ample additional hypotheses available for examination stemming from this project.

Major lessons learned

- Brown treecreeper short-term survival was very high, and social groups maintained high group cohesiveness, suggesting that the species handled the translocation process well. This result emphasized the importance of knowledge of the social groups present in the source population.
- Regardless of existing knowledge of brown treecreeper habitat preferences in other populations, re-introduced individuals selected habitat contrary to expectations, selecting forest rather than eucalypt woodland. This result emphasized that behavior and habitat use information from prior studies within a source population may not approximate that which is observed within a re-introduced population.
- Brown treecreepers displayed an increased probability of foraging on a log or on the ground when within areas that coarse woody debris had been experimentally added to the reserves. This behavior indicated that the addition of coarse woody debris benefited the species potentially through influencing foraging efficiency, and also demonstrated the value of using behavior as a bio -indicator for restoration success.
- Further consideration of, and investigation into, the elements influencing predation by natural predators is required to enable conservation of ground-foraging insectivores and is essential prior to any re-introduction of similar species.

- Continued restoration is required in the reserves where the re-introduction took place. This includes consideration of finer-scale habitat components such as management of the ground layer through promoting the development of a cryptogamic crust, an increased leaf litter layer, reduced weed cover and controlled levels of grazing pressure by both native and exotic herbivores. This result reinforces the need to closely examine the habitat suitability before a translocation.

Success of project

Highly Successful	Successful	Partially Successful	**Failure**
			√

Reason(s) for success/failure:

- The re-introduction process itself was highly successful particularly given the high survival rates over 24 hours and 3 days post-release (93% and 91% respectively) and the high level of group cohesion observed immediately after release.
- Although the criteria for success regarding adult survival over 3 days (70% survival) and 4 weeks (50% survival) were achieved, the survival rate of adult brown treecreepers over 1 year (15%) did not meet the criteria for success (40% survival). This was despite attempts to address the previous causes for the decline for the species and restore the habitat. This indicates that in order to establish a viable population, further releases of larger numbers of individuals over several years would be required.
- No instances of successful reproduction was observed despite regular monitoring of females. Although, a male was seen feeding a female on seven separate occasions during November 2010. This was noteworthy given that breeding females appear not to accept courtship feeding unless they intend to attempt to reproduce.
- Although in terms of survival rates, the re-introduction did not succeed, the project provided unique and important knowledge regarding brown treecreeper behavior, habitat use, movement, restoration ecology of eucalypt woodlands, and the procedures of re-introducing a social bird species.

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Re-introduction of the western bluebird to oak-prairie habitats in Pacific Northwest, USA

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Introduction

The Western bluebird (*Sialia mexicana*) occurs throughout much of western North America and breeds in a variety of open habitats where nest cavities, low perches, and an open understory are present. In the Pacific Northwest, west of the Cascade crest, the northern edge of the bluebird's range has undergone large -scale retraction due to the loss, fragmentation, and degradation of oak-prairie habitats, where they most commonly occur (Altman, 2011). Coastal mainland breeding populations of bluebirds disappeared from British Columbia, Canada and Washington, USA in the 1970s, while island populations in the San Juan and Gulf Islands archipelagos began disappearing in the 1960s. The last observed breeding in the region occurred in 1995 on Vancouver Island, BC. Because the species occupies a broad array of open habitats, the primary cause of their decline was apparently the loss of a critical habitat element, nesting cavities. In some areas, such as in the Willamette Valley, OR and south Puget Sound, WA, nearly extirpated populations have recovered following the establishment of nestboxes programs. This case study reports on the 6-year (2007 - 2012; 5 yrs. translocation, 1 yr. post-translocation) re-introduction of the bluebird to San Juan Island, WA.


Western bluebird male in aviary © Gary Slater

Goals

- Goal 1: Capture >90 wild adults from donor sites and safely transport and release on San Juan Island during a 5-year period.
- Goal 2: Monitor reproduction and survival rates in the re-introduced population to evaluate translocation methods and re-introduction success.
- Goal 3: Establish a self-sustaining breeding population on San Juan Island and adjacent islands.

- Goal 4: Use the bluebird as a flagship emblem for oak-prairie conservation.

Success Indicators

- Indicator 1: Released individuals and their offspring breed successfully and reestablish migratory pathways between wintering grounds and the re-introduction site.
- Indicator 2: Population size increases annually.
- Indicator 3: Demographic measures (reproduction and survival) in the re-introduced population are similar to other Pacific Northwest populations.

Project Summary

Feasibility: The re-introduction of bluebirds to San Juan Island was considered appropriate and timely for several reasons. First, the likelihood of bluebirds re-establishing a population on San Juan Island without assistance appeared low. The long distance (165 km) and large area of unsuitable habitat (i.e., urban Seattle and Puget Sound) between San Juan Island and the closest source population (south Puget Sound) apparently hindered dispersal, because there was no evidence of successful colonization in the three decades since the species was extirpated, even though the source population showed substantial growth. Second, a pre-project assessment indicated that sufficient habitat was available in north Puget Sound, centered on San Juan Island, to support a bluebird population. Local conservation organizations (e.g., San Juan Preservation Trust, San Juan County Audubon Society) promoted the protection and restoration of the prairie-oak ecosystem, ensuring that habitat would be available in the future. Third, the cause of their extirpation was considered to be the loss of a particular habitat element, cavities for nesting, rather than a more complex set of issues unable to be addressed through management. Nest boxes have been used as management tool to increase the availability of cavities for many cavity-nesting species and they have played a critical role in the recovery of Eastern and Western bluebird populations in many parts of North America. Local conservation partners encouraged the placement of nestboxes, and over 500 were established during the course of the project. Finally, successful translocation methodologies had been developed for Eastern bluebirds in Florida and these methodologies were believed to be transferrable to a re-introduction of Western bluebirds (Slater, 2001).

Implementation: We translocated bluebirds to San Juan Island in each breeding season (March - June) from 2007 to 2011. The source population was Joint Base Lewis-McChord Military Base, approximately 165 km from the re-introduction site; several pairs were translocated from Oregon (450 km away). Most translocations involved breeding pairs, although we moved some pairs with dependent young later in the breeding season. In 2010 - 2011, we translocated a few single females because we observed a higher ratio of males to females in the re-introduced population. At the release site, bluebirds were placed in outdoor aviaries, which allowed open views, yet provided protection from the elements. Aviaries contained multiple perch choices, a nest box for roosting, and food (mealworms and crickets) and water *ad libitum*. Initial releases were conducted in the San Juan Valley, which historically held the most oak habitat on the island.

Release sites were selected based on the presence of suitable habitat (e.g., proximity to oaks, appropriate foraging habitat), the willingness of landowners to host an aviary and place nest boxes on their property, and, upon establishment, the proximity of bluebird territories. Release sites for single females were selected based on the presence of a single territorial male.

We captured and translocated 102 adults and 35 juveniles; 2 adults and 1 juvenile died in the aviary, but the remaining were released in good condition. In 2007, we placed 8 adult pairs in 1 m x 1 m x 2 m aviaries (small), releasing them after 4 - 5 days. We discontinued this strategy following low establishment (only 1 pair) and high rate of dispersal (45%) back to the source population. In the following 4 years, we placed breeding pairs, captured early in the breeding season, in 2 m x 2 m x 2 m aviaries (large), holding them for 1 - 3 weeks. Twenty seven of 65 (42%) individuals released as pairs (one with a resident bird) established a breeding territory. Pairs translocated with dependent young (10 - 12 days old) were placed in a large aviary (the young in a nestbox) and were released 1 - 10 days after nestlings fledged. Six of 15 (40%) adults established a breeding territory; 7 of 35 (20%) juveniles returned the following year to breed. Single females were placed in a small aviary and released after 3 - 5 days in the presence of a free-living male; 3 of 5 (60%) single females established a breeding territory. On all established territories, we provided supplemental food (mealworms) to birds during periods of cool (<16° C), windy, and rainy weather and when pairs were feeding nestlings.

Post-release monitoring: We found evidence of successful breeding in each year of the project and both translocated individuals and their locally-produced offspring reproduced successfully. Annual counts of adults indicated that the re-introduced bluebird population grew in each year of the project during the translocation period, and at the end of the 2011 breeding season the minimum estimate of population size was 38 individuals (14 breeding territories).

Constructing aviary © Gary Slater

In 2012, we found fewer individuals, but there were still 14 breeding territories. From 2007-2012, we monitored 87 nests, which fledged 274 juveniles. Fecundity and survival estimates in the re-introduced population did not differ significantly from reference populations in the Pacific Northwest (Keyser *et al.*, 2004; Kozma & Kroll, 2010).


Typical habitat in release area © Gary Slater

Major difficulties faced

- In 2007, our attempt to use a smaller, and easier to move, aviary and a shorter holding period proved ineffective, and thus we returned to larger aviaries and longer holding periods, techniques used for Eastern bluebirds.
- Initially, annual return rates for juvenile males were higher than females producing a male-biased sex ratio in the nascent population.
- Nest predation by house sparrows and other mammals is a leading factor in nest failures.
- Poor reproduction in 2011 and 2012, due to unusually cold and rainy breeding seasons, is a significant concern to this small and vulnerable population.

Major lessons learned

- Holding bluebird pairs for longer periods (1 - 3 weeks) in large aviaries appeared more effective than short holding periods (3 - 5 days) in small aviaries.
- Breeding pairs captured earlier in the breeding season (before mean incubation date) were more likely to establish a territory than pairs captured later in the breeding season.
- Similarly, translocating and releasing pairs with juveniles earlier in the season to allow pairs time to re-nest was more successful than later releases.
- Releasing family groups when young are 2 - 4 days old appears to reduce dispersal from the release site, although aviary sites need to include patches of shrubby vegetation to provide cover for juveniles.
- Translocations of single females was highly effective and thus provides evidence of a technique to successfully address biased sex ratios in small re-introduced populations.
- In contrast to the re-introduction of Eastern bluebirds in South Florida, paired individuals typically maintained pair bonds, providing support for translocating pairs rather than single individuals.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- A dedicated partnership of conservation groups that provided the full spectrum of expertise, from administration to technical to local knowledge, necessary for a successful re-introduction project.
- The ability to adapt and modify translocation strategies during the project.
- Participation by local conservation organizations, San Juan Preservation Trust and San Juan Audubon Society, who actively engaged the local community in participating and supporting the re-introduction project.
- The presence of a large donor population, which allowed us to reach our target release number within our proposed timeframe.
- While we successfully established a small population on San Juan Island, further monitoring will be required to evaluate population persistence and determine whether the re-introduction can be considered "successful".
- The success of the re-introduction effort on San Juan Island spurred the expansion of the project to Vancouver Island, 25 miles away. The creation of another local population should increase the likelihood of long-term persistence for the regional population.

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Re-introduction and recovery of the red wolf in the southeastern USA.

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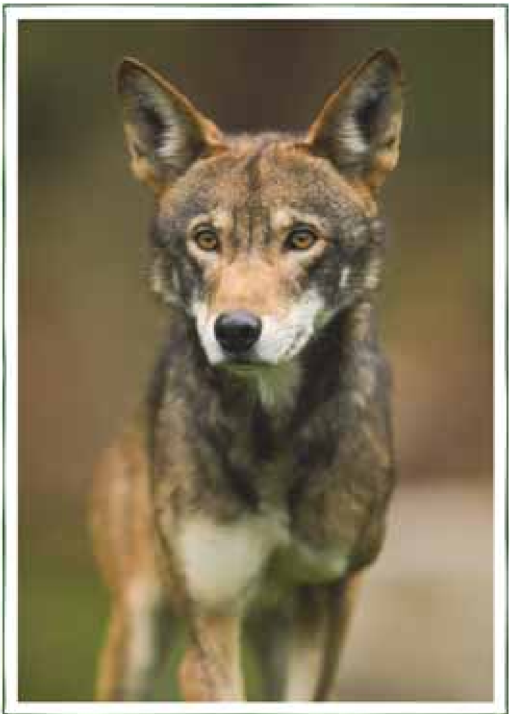
Introduction

The red wolf (*Canis rufus*) is one of the Earth's most imperiled canids. Once occurring throughout the southeastern USA, red wolves were decimated by predator-control programs and habitat degradation. Remnant populations of red wolves were further threatened by hybridization with expanding coyote (*C. latrans* var.) populations. To protect the red wolf from extinction, the U.S. Fish and Wildlife Service (USFWS) extirpated the red wolf in the wild and established an *ex situ* breeding program with plans to restore the species to a portion of its former range. Only 14 individuals would reproduce to become the founding ancestors of all red wolves existing today. Successful *ex situ* reproduction prompted a re-introduction of red wolves in northeastern North Carolina (NENC) in 1987. A second re-introduction was initiated in 1991 in the Great Smoky Mountains National Park, but later terminated because of disease and low pup survival. The restored population of red wolves in NENC has expanded to included 90 - 110 wolves occurring over more than 6,000 km^2. Nearly 200 red wolves are maintained in more than 40 zoos/ nature centers throughout the USA. The red wolf is federally listed as Endangered under the Endangered Species Act (ESA) and IUCN Critically Endangered D (IUCN, 2012).



Red wolf © John Froschauer/PDZA

Goals

Goals and success indicators of the Red Wolf Recovery Program are taken from the Red Wolf Recovery/Species Survival Plan (USFWS, 1990) and the Red Wolf 5-Year Status Review (USFWS, 2007).

- Goal: Achieve a series of geographically independent populations of red wolves, through re-introduction, that are numerically large enough to have the potential for allowing natural evolutionary processes to work within the species (USFWS, 1990).

Fostering of red wolves © David Rabon/USFWS

Success Indicators

- Indicator 1: Establish and maintain a series of at least three red wolf populations via restoration projects within the historic range of the species. Each population should be numerically large enough to have the potential for allowing natural evolutionary processes to work within the species. This must be paralleled by the cooperation and assistance of at least 30 captive-breeding facilities.
- Indicator 2: Preserve 80% to 90% of the genetic diversity found in the founding population (14 individuals) of red wolves for a period of 150 years or more.
- Indicator 3: Remove those threats that have the potential to bring about extinction of the red wolf. Achieving this objective will include maintaining a total wild population of at least 220 wolves and a captive population of approximately 330 wolves.
- Indicator 4: Maintain the red wolf in perpetuity through embryo banking and cryogenic preservation of sperm.

Project Summary

Feasibility: With the passage of the ESA, a red wolf recovery program was established in partnership with Point Defiance Zoo and Aquarium (WA, USA), to coordinate captive breeding efforts and determine "pure" red wolves as part of a planned extirpation of red wolves in the wild. Between 1973 and 1980, only 14 canids captured within the remaining red wolf range were determined to be pure red wolves and successfully bred in captivity. By 1980, the red wolf was extirpated in the wild. In 1984, the captive breeding program was accepted by the Association of Zoos and Aquariums for development of a Species Survival Plan (SSP), and plans for re-introduction were initiated. To assess various restoration approaches (e.g., acclimation, release, and recapture techniques), captive-born red wolves were released on an island propagation site. After public opposition at the initial re-introduction, Alligator River National Wildlife Refuge (ARNWR, NC) was chosen as the red wolf re-introduction area. Extensive analyses determined feasibility due to the absence of coyotes, the lack of livestock operations, and availability of prey species. To garner public support, traditional recreational activities, such as hunting and fishing, were allowed to continue within the re-introduction area, and the re-introduced population was designated "non-essential experimental" under Section 10(j) of the ESA.

Implementation:
Captive breeding: With very small populations, survival can be affected by genetic drift (random loss of allele frequency) and inbreeding depression. Currently, gene diversity in the captive red wolf population is approximately 89.65% of the founder population (N = 14), and there is little evidence of inbreeding depression.
Pre-release conditioning: As a strategy to propagate wild red wolf offspring for release, breeding pairs of wolves from captive breeding facilities were relocated to several island propagation sites. The wolves were released on the islands to live, hunt, breed, and raise their young in a natural, albeit space limited, environment. Their offspring, having been raised "wild," would be relocated to the mainland re-introduction site when they reached dispersal or reproductive age. The concept being that wild-raised red wolves would have learned to hunt and live as wild animals and were more likely to survive following release than captive -reared wolves. Most of these sites were discontinued due to human-wolf interactions or funding constraints. Currently, the St. Vincent National Wildlife Refuge (FL, USA) island propagation site remains operational.
Re-introduction: In 1987, after briefings to the public, state, and other federal agencies, re-introduction efforts of the red wolf began on ARNWR with the release of four captive-born, male-female wolf pairs. The captive-born wolves were housed in 225 m^2 acclimation pens prior to release. During acclimation, human contact was minimized, and feeding regimes were altered to resemble wild conditions. Prior to release, the wolves were given a health check, vaccinated, treated for parasites, weighed, and fitted with VHF radio-telemetry collars (Phillips *et al.*, 2003). To encourage the wolves to remain near the release site, and to facilitate predatory diet and habits, the wolves were provided supplemental food (generally deer carcasses) for 1 - 2 months following release.
Fostering: The insertion of captive-born wolves in a wild-born litter has been a successful tool to increase the number of wild red wolves and enhance the genetic diversity of the re-introduced population. Typically, captive-born pups are inserted into a wild litter when the recipient and donor pups are between 10 - 14 days old, and the recipient litter is small enough to accommodate the additional litter mates.
Adaptive management: The expansion of the coyote into the red wolf recovery area has resulted in interbreeding and coyote gene introgression into the wild red wolf population. To reduce hybridization, an adaptive management plan was developed that uses sterilized, hormonally-intact (via vasectomy and tubal ligation) coyotes as territorial "placeholders." The "placeholder" coyotes will not interbreed with red wolves, and they exclude other coyotes from their territory. Ultimately, the "placeholder" coyotes are replaced by red wolves either naturally (e.g. displacement) or via management actions (e.g., removal followed by insertion or natural dispersal of wolves into the territory).

Post-release monitoring:
Population estimation: Adult and juvenile red wolves are live-trapped, fitted with VHF radio-telemetry collars, and monitored several times a week from fixed-wing aircraft and ground surveys. Radio-telemetry techniques determine wolf movements, territory usage, pairings and interactions, den establishment and location, and fates of individuals. Passive integrated transponder (PIT) tags are

implanted in all pups found during the spring den search for identification when later captured and radio-collared as adults. Population estimates of wild red wolves are calculated by adding the number of actively monitored radio-collared wolves and PIT-tagged pups recorded during the spring whelping season.
Current conservation status: As of 2012, ~90 - 110 red wolves are surviving in the wild, of which 65 are regularly monitored through radio-telemetry. However, the species only exists in the wild in the one re-introduced population. The captive breeding population remains stable at ~200 red wolves.

Major difficulties faced

- *Human-caused mortality*: From September 1987 through December 2012, mortalities were documented for 364 wild red wolves in NENC. Demographic data were available for 357 red wolves, including 72 pups, 94 juveniles, and 191 adults (many of which were breeders). For the first 25 years of re-introduction, causes of death were determined for all red wolves in the NENC re-introduction area. Causes of red wolf mortalities included suspected illegal activities, involving gunshot, poisoning, and other suspected illegal take (30%); vehicle collisions (20%); health-related causes (16%); intraspecific competition (6.5%); management actions (5.0%); private trapping (3.5%); and, unknown causes (19%). Fifty-seven percent of all observed mortalities (72% of mortalities with a known cause of death) during this period were human-caused and potentially avoidable. During the past nine years (2004 - 2012), the average annual number of gunshot-caused mortalities has increased ~375% when compared to earlier years (1988-2003).
- *Disease:* Canid diseases have threatened both re-introduced and captive red wolf populations. The magnitude of risk to the red wolf species overall is partly offset by captive red wolves held in more than 40 SSP zoos and nature centers across the USA. Risk of disease is also partly offset by intensive vaccination programs for both re-introduced and captive red wolves. However, veterinary research scientists caution against the assumption that vaccinated red wolves are adequately protected against diseases. The diseases of greatest concern are canine distemper (Genus *Morbillivirus; CDV)*, canine parvovirus (Genus *Parvovirus; CPV1, CPV2)*, leptospirosis (Genus *leptospira*), hemobartonellosis (*Haemobartonella canis*), borrelliosis (Lyme disease, *Borrelia sp.*), demodectic mange (*Demodex canis* mites), sarcoptic mange (*Sarcoptes scabiei* mites), heart worm (*Dirofilaria immitis*), and rabies (Genus *Lyssavirus, rabies virus*). The impacts of CPV2 parvovirus on pup survival in the Great Smoky Mountains National Park re-introduction area eventually contributed to the termination of that project. Fortunately, to date, none of these diseases have occurred at sufficiently high levels to cause an epidemic. However, mange (N = 17) and heartworm (N = 7) have been confirmed as repeated sources of red wolf mortality in the re-introduced population. New threats also are becoming more prevalent in local domestic dogs, including the Lyme disease-causing bacteria *Borrelia burgdoferi*.
- *Interbreeding with coyotes:* The recovery and restoration of red wolves requires the careful management of coyotes and occasionally red wolf-coyote hybrids in the red wolf re-introduction area. The non-native coyotes spread across the eastern USA, reaching NENC in the early to 1990s. It soon was

recognized that interbreeding between red wolves and coyotes would produce hybrid offspring resulting in coyote gene introgression into the wild red wolf population, and that this introgression would threaten the restoration of red wolves. An adaptive management plan (Rabon *et al.*, 2013) was developed to reduce interbreeding and introgression while simultaneously building the red wolf population. The adaptive management plan effectively uses techniques to capture and sterilize hormonally intact coyotes via vasectomy or tubal ligation, then releases the sterile canid at its place of capture to act as a territorial "placeholder" until the animal is replaced by wild red wolves. Sterile coyotes are not capable of breeding with other coyotes, effectively limiting the growth of the coyote population, nor are they capable of interbreeding with wild red wolves, limiting hybridization events. In addition, the sterile canid will exclude other coyotes from its territory. Ultimately, the placeholder coyotes are replaced by the larger red wolves either naturally by displacing the coyote or via management actions (e.g., removal of the coyote followed by insertion of wild or translocated wolves).

Typical red wolf habitat © Melissa McGaw

- *Climate change and stochastic events:* Natural weather events and global climate change will play growing roles in long-term survival and recovery of red wolves, especially in the re-introduced red wolf population. The re-introduced wild red wolf population in NENC, as well as many of the captive SSP facilities, is subject to the adverse effects of annual tropical storm activity. Hurricane Isabel (2003) resulted in the deaths of two captive red wolves, and Hurricane Sandy (2012) resulted in the death of one captive red wolf. Although there has been no noticeable long-term impacts observed on the red wolves in the re-introduced population, the red wolf restoration area and associated habitats and prey species are vulnerable to sea level rise and flooding related to climate change and tropical events. Additional long-term changes in habitat availability, prey abundance, and other ecological or landscape factors will occur with climate change. Thus, long-term assessment and planning are needed that consider the current re-introduced and future populations in the context of tropical storm activity, global climate change, and resulting changes in the North American landscape over time.

Major lessons learned

- *Partnerships and cooperation are essential for success:* Cooperation and creative partnerships are essential to the success of any re-introduction program. The successful captive red wolf breeding program is a result of the cooperation of Point Defiance Zoo and Aquarium, the development of an Association of Zoos and Aquariums SSP program, and the participation of numerous SSP-affiliated zoos and nature centers. Consequently, captive breeding has become a foundation of the success of the red wolf recovery program. Researchers and other science-based partners have provided data and information necessary to make management decisions that support restoration actions to ensure sound conservation approaches to recovering the red wolf. Re-introduction requires cooperation and partnerships among many diverse groups, particularly among local, regional, and state-wide governments, private landowners and land managers, researchers, and special-use groups and organizations.
- *Disease prevention and surveillance are prudent:* Because canid diseases can spread quickly, they can cause serious setbacks in red wolf recovery, and remain serious threats to all red wolf populations. As evidenced by the termination of the Great Smoky Mountains National Park re-introduction project due to poor pup survival and parvovirus, additional precautions are needed to proactively address potential disease outbreaks in any re-introduced red wolf population. The establishment of at least two more re-introduction sites within red wolf historic range could partly alleviate disease risk. The import of existing and new strains of canid diseases carried into a re-introduced red wolf population also is a concern. Domestic hunting dogs and imported coyotes from elsewhere in USA are two potential outside sources of disease. A red wolf disease prevention and surveillance program has been recommended to ensure long-term survival for any red wolf re-introduced population.
- *Early development of regulatory mechanism is critical:* The red wolf remains federally listed as Endangered throughout its historic range. However, the red wolf was declared extinct in the wild in 1980 when the last known remaining red wolves were brought into captivity. Therefore, red wolves in captivity are listed Endangered, whereas re-introduced red wolves are designated as a nonessential experimental population under 10(j) of the ESA.
- The nonessential experimental status for the

Red wolf with radio-collar © Ryan Nordsven/USFWS re-introduced population of the red wolf is a helpful mechanism which allows managers to work cooperatively with partners to enhance red wolf recovery and resolve problems. The nonessential experimental status also allows flexibility for landowners and managers, and other citizens by allowing exceptions to the prohibitions of take under the ESA when a red wolf constitutes a demonstrable threat to human safety or livestock, provided it has not been possible to eliminate such threat by live capture and relocation of the wolf. Such flexibility allows less regulation while addressing needs in human safety and property. However, there is room for improvement to ensure that federal listing status of the red wolf is mirrored by state listing status such that it promotes red wolf conservation and synergy in red wolf recovery.

- *Sterile coyote placeholders can deter hybridization:* During the initial site selection process for the red wolf re-introduction program, the NENC red wolf recovery area was considered uninhabited by coyotes. However, coyotes have expanded their historical range eastward; individuals were observed in the recovery area beginning in the early-1990s. As a result, an adaptive management plan was needed to attempt to eliminate the threat of hybridization. Research has demonstrated that sterilized coyotes remain territorial and continue to defend space. It is this concept of holding space that is being applied to manage hybridization by providing managers time, information, and a higher degree of control over the recovery landscape, while simultaneously providing reproductive advantage to the red wolf. Ultimately, sterilization is a method that allows territorial space to be held until that animal can be replaced naturally or by management actions. Sterile or “placeholder” coyotes are then naturally replaced when the larger red wolves displace or kill the coyote. Occasionally, a coyote may be removed from an area when there is an opportunity to insert a wild or translocated red wolf into that territory or if there is a red wolf dispersing into that area.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- *Socio-politics:* Socio-political views of the red wolf, and wolves in general, have a long history. The wolf was maligned in folktales, fables, and fairy tales, and persecuted in reality. Euro-American colonists, acting on prejudice, established a bounty on the wolf that spread like an epidemic with the growing nation. With the expansion and increasing number of pastoralists, the wolf was seen as much as an ecological competitor that threatened livestock and livelihoods as it was a diabolical and malevolent beast. The widespread use of a bounty extirpated the wolf in many regions. By the early-1900s, government-operated predator control programs had the task of systematically exterminating the wolf, further driving the red wolf to near extinction. Eventually, the wolf was romanticized in literature, reversing the public’s sentiment, or at least the government’s role in their eradication. By the later part of the 20th century the public’s attitude had swayed enough to support legislation to protect the species. Decades of successful restoration activities and the unfulfilled prophecies of the wolf's devastating impact on the local wildlife or livelihoods has assisted in furthering the positive change in attitudes. But localized animosity toward the red wolf still exists in landowners and land managers that see the wolf as an ecological competitor despite the increasing recognition of red wolves as ecologically important. The red wolf continues to be persecuted, requiring additional management interactions to maintain the red wolf population.

- *Adaptive management techniques:* Adaptive management techniques have shown that sterilization is a method that allows territorial space to be held until that animal can be replaced naturally or by additional management actions. Sterile or "placeholder" coyotes are then naturally replaced when the larger red wolves displace or kill the coyote. Ongoing analyses suggest that red wolves always win over coyotes in the battle of territorial disputes, whether management actions were taken or not to remove a coyote. Preliminary data analyses show no instances of a coyote successfully defending a territory against a red wolf. Space is limited in the re-introduction area. Ideally, within the re-introduced red wolf population in NENC, that space is initially best occupied by breeding pairs of red wolves, non-breeding mixed (red wolf/ coyote) pairs, and non-breeding coyote pairs. By sterilizing coyotes, introgression of non-wolf genes will be controlled and territories will be unavailable for colonization by breeding coyote pairs or breeding red wolf-coyote pairs. In addition to the ~65+ radio-collared red wolves, there are also ~60+ sterilized, radio-collared coyotes regularly monitored. As the red wolf population grows, having space available for dispersing red wolves will become increasingly important, and this space will be provided through natural interspecific competition and/or management actions.
- *Persistence and patience:* Coyote expansion and the threat of hybridization and genetic swamping of the small remnant red wolf populations ultimately lead to an abandonment of the attempt to preserve the red wolf in the wild in the late-1960s. When planned extirpation of the wild red wolves and the establishment of a captive breeding program were determined to be the only solutions, then the captive breeding process was marred by the availability of pure red wolves. Only 14 red wolves were determined "pure" and verified through a breeding certification program, becoming the founding population of all red wolves in existence today. Captive breeding also was hampered by a slow start in the production of viable offspring. It was not until 1977 when the first litter of red wolves was born in captivity that the real steps in red wolf recovery were made. The red wolf captive breeding program has grown and developed since 1973, ensuring and maintaining the genetic diversity of the species. More than 40 zoos and nature centers that breed red wolves have committed substantial resources, without compensation, to the captive breeding effort. Re-introduction of captive-born red wolves into the wild began in 1987 and continued until 1994. However, red wolves were born in the wild every year since the first wild-born litter in 1988. Fortunately, program partners remain committed to re-introduction, and new partners are joining the recovery effort.

- *Continued research and monitoring:* An adaptive approach to recovery program management has allowed research to address ongoing and emerging issues facing the red wolf recovery program. Several identified research needs include demographic analysis of the effects of gunshot mortality on red wolf population dynamics; development of a two-species model to show how the presence and interactions with coyotes impact habitat suitability indices and red wolf carrying capacity; monitoring how pre-release conditioning can improve survival and establishment of a territory; careful genetic management of the captive population and the development of artificial insemination and cryopreservation techniques; and investigating how pup fostering can increase numbers and genetic diversity in the re-introduced population.

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Re-introduction of the Mexican wolf in the Sierra Madre Occidental, Mexico

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Introduction

Mexican wolves were extirpated from the wild in the late 1970s, since then a captive breeding program involving the United States and Mexico has resulted in a bi-national recovery program with a current captive population of 253 (June, 2013). A wild population of 75 (December, 2012) in the Blue Range, US is the result of a re-introduction program started by the USFWS in 1998. Since 2006, several stakeholders in Mexico began a re-introduction process that culminated with the release of 10 Mexican wolves during 2011, 2012 and 2013, representing three family groups. Survivorship of the wolves has been affected by poisoning livestock carcasses and negative attitudes toward the re-introduction. A single stakeholder approach and supplemental feeding has increased the survivorship of those individuals released in 2013. We believe the process to establish a self sustaining population is just beginning, but there are changes occurring that might allow to achieve this in the mid term.

Goals

- Goal 1: Identification of re-introduction sites within the historical species range in Mexico.
- Goal 2: Constant monitoring of released individuals in the wild.
- Goal 3: Constant communication with local stakeholders to assess their perception of the project and apply adaptive management practices.

Success Indicators

- Indicator 1: Establishment of one or more self sustaining Mexican wolf populations in Mexico.
- Indicator 2: Increase Mexican wolf survivorship in the wild.
- Indicator 3: Achieve Mexican wolf reproduction in the wild.

Project Summary

Feasibility: A captive Mexican wolf program was established in the late 1970s, with the capture of five wild individuals in the Mexican States of Durango and Chihuahua. Currently, the program has the support of 52 institutions in Mexico and the United States and is composed by 253 individuals (June, 2013). The program is represented by three genetic lineages incorporated in the last 30 years. In 1998 the United States Fish and Wildlife Service (USFWS) re-introduced the species in the Blue Range Area (Arizona and New Mexico States), to date that population has reached 75 individuals. Parallel to the efforts carried out in the US, Mexico first developed a Mexican Wolf recovery project in 1999. In 2007 a recovery strategy for endangered species (PROCER) was established and a recovery plan for each one developed, included the Mexican Wolf (PACE: Lobo Gris Mexicano). In 2006, a group of stakeholders (including scientists, government and nonprofits) assessed the feasibility of carrying out the first re-introduction of Mexican wolves in Mexico. Six areas were selected (based on landscape suitability) to be surveyed for prey abundance (particularly ungulates) and social attitudes towards wolves. In 2008 two areas were selected as possible candidate areas both in the Sierra Madre Occidental: Sonora and Chihuahua. The land tenure landscape of Mexico, contrary to the United States, is dominated by privately owned lands (single individuals or community owned land "ejidos") with minimal or no federally owned land, which implies that in order to release individuals on the ground we need the written approval of the landowner.

Mexican wolf in the wild

Implementation: The first group of Mexican wolves selected for re-introduction was rehabilitated for hunting abilities, other behavioral attributes and social cohesion. The first re-introduction occurred in Sonora, with a family group that included 2 males and 3 females (1 female and her 4 and 5 year old offspring). Additionally a male was released in the same area 6 months later to pair with the surviving female. The second re-introduction took place in Chihuahua in October 2012 and comprised a pair (6 year old male with a 5 year old female). A second release in the same area in April 2013 included another pair (3 year old male with a 7 year old female). All individuals were fitted with a satellite radio transmitter to obtain telemetry locations to determine their movement and survivorship. The areas selected for release had a different social approach; the first (Sonora) included a series of conversations with livestock producers with the aim of exposing the possible benefits of ecological restoration resulting from wolf presence and the impact observed in other experience. The second area (Chihuahua) had a less publicized approach that included custom-made talks with individual stakeholders resulting in the written consent of them accepting the release of wolves on their land. In the first stage of the release program (2011-2012), there was an allowance (economic support) for local

Mexican wolf at release site

landowners involved in the project provided by the National Commission of Natural Protected Areas (CONANP-SEMARNAT). All the monitoring effort and release activities (2011 - 2013) were implemented by academic and non-profit institutions, also through a grant by CONANP.

Post-release monitoring: Mexican wolves were monitored via satellite and ground telemetry. The satellite telemetry was carried out in collaboration with the USFWS, using the same programing of the collars as it has been used in the Blue Range Wolf Recovery Area (3 to 4 locations per day, with locations obtained between 2 to 4 days apart). Ground telemetry depended on the topography and safety conditions for the technicians. The first released family group split into two entities, a single 5 year old female and the other four remaining together. These four individuals were found poisoned during the 1 to 2 months after release. After almost 6 months of territorial stability in the region, the single female began a major movement that ended almost 200 km south of the release site, where her signal disappeared seven and a half months after the release.

During the releases in the second area (Chihuahua) two elements to favor Mexican wolf adaptation and monitoring were added: food supplementation and camera traps were placed in the release sites. The first pair released in Chihuahua had a bond and remained as such; their monitoring has been facilitated by this behavior. These individuals have fed on white tailed deer, cottontail rabbits, small peccaries and livestock carcasses. The second pair released in Chihuahua did not present such bonding structure and resulted in an immediate separation, which has resulted in a complex monitoring pattern; they have not settled and established a definitive home range. The male has traveled extensively north and the female has traveled south, both have dispersed on average 40 km of the release site. The habitat used by these two are significantly different, the male using Chihuahuan desert flatlands, the female has remained associated to high elevation pine-oak forests.

Major difficulties faced

- Livestock producer's antagonistic behavior in Sonora resulted in low survivorship of released individuals.

- Poor husbandry in livestock operations provide carrion to Mexican wolves, both in Sonora and Chihuahua.
- These problems have lowered the success of the re-introduction process.
- Local ego's has been an obstacle to successful communication of the project results.

Major lessons learned

- Mexican wolves are capable of living in a privately owned dominated landscape, supplemental feeding has been a major tool to facilitate and acclimatize the individuals to their environment. Food habits analysis have shown the use of native ungulates, and to some degree livestock carcasses.
- Livestock carcasses are readily available to Mexican wolves and other predators which are perceived as "depredation events" into the eyes of ranchers and livestock producers, alternative management should result in lower availability of carcasses The social approach in these projects should consider primarily rural areas, where the direct contact and information is needed, supported with programs like livestock insurance, support in management and incentives to conservation.
- The program requires long-term support of the Federal and State governments and strong collaboration with academics and NGO.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- An individually social based approach to the program has facilitated the tolerance of wolves in the second re-introduction site. Contrary to the first re-introduction site.
- Alternative husbandry techniques should be implemented by local stakeholders that have shown acceptance to the program, resulting in neighboring ranches implementing those techniques.
- Additional availability of Mexican wolves should favor the survivorship of more individuals.

Re-introduction of the Iberian lynx, Andalusia, Spain

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Introduction

The Iberian lynx (*Lynx pardinus*) is an endemic felid of the Iberian Peninsula. Once widespread across the Iberian territory, only about 100 Iberian individuals were found to occur in 2002 into two isolated populations in Andalusia (Southern Spain): Andújar-Cardeña and Doñana (Guzmán *et al.*, 2004). Given this critical situation, the Iberian lynx was the only felid species catalogued by IUCN as "Critically Endangered" in 2003 (IUCN 2003). Starting in 2002, three consecutive EU-funded Life conservation projects (summarizing 14 years) are being developed by the Andalusian Regional Government of Environment in order to stop the decline of the population and restore extinct populations through re-introduction (Simón *et al.*, 2012). Moreover, a captive-breeding program was initiated in 2004 with the main goal of providing individuals to be released in re-introduction programs (Vargas *et al.*, 2008). The first re-introduction program began in 2006 with the selection of optimal areas for re-introduction. Two areas close to Andújar-Cardeña nucleus were selected for re-introduction: Guadalmellato and Guarrizas. Releases began in 2009. Currently, 34 individuals have been re-introduced in the wild. Demographic parameters of both re-introduced populations show a positive evolution, and the total wild Iberian lynx population in 2012 was estimated to be 320 individuals.

Iberian lynx © Manuel Moral

Goals

- Goal 1: Decrease Iberian lynx extinction risk through the creation


Typical Iberian lynx habitat © Miguel Simon

of two connected re-introduced nuclei that would strengthen the remnant population.
- Goal 2: Identification of optimal areas (regarding to habitat, resources and threats) for the re-introduction within the former Iberian lynx range in Andalusia.
- Goal 3: Correction of main limiting factors through habitat management, resource improvement, reduction of threats and obtaining a strong social support for re-introduction.
- Goal 4: Provide enough individuals (both wild-born and captive-born) to create two self-sustaining populations (connected to each other), in 15 years.
- Goal 5: Evaluate effectiveness of all processes in order to establish useful re-introduction protocols for the species.

Success Indicators

- Indicator 1: Establishment of 15 breeding females per area after 7 years of releases.
- Indicator 2: Establishment of 30 breeding females per area after 15 years of releases.
- Indicator 3: Achieve annual survival rates higher than 50% in released individuals.
- Indicator 4: Obtain the interconnection of areas of presence with areas of re-introduction to reach an only meta-population.
- Indicator 5: Down-list the IUCN threat category of the Iberian lynx after 10 years of releases.

Project Summary

Feasibility: The Iberian lynx need well-preserved Mediterranean scrubland and high wild rabbit (its staple prey) densities. Habitat transformation (mainly due to both forestry plantations and infrastructure development) has provoked suitable habitat to be restricted to hunting private lands. Thus, collaborative agreements with owners and hunting societies are essential in Iberian lynx conservation (Simón *et al.*, 2012, 2013). After a careful selection and a correction of limiting factors in optimal re-introduction areas in Andalusia, and once a strong social support was guaranteed, releases began in early 2010. The re-introduction program was designed by a multidisciplinary team and approved in an international seminar. All the stages of the program are being covered according the planned agenda. Up to January 2013, 19 (8 males:11 females) and 15 (7 males:8 females) Iberian lynxes in Guadalmellato and Guarrizas were released respectively. Out of the total released lynxes, 50% were wild-caught individuals and 50% captive-bred ones. Collaborative agreements and public awareness are being important to count on the local population support of the program.

Implementation: Prior to releases, wild rabbit populations were enhanced through habitat management. Moreover, a sanitary surveillance program was implemented in wildlife in order to evaluate sanitary risks for re-introduced individuals. Meanwhile, the main potential threats were decreased through awareness and poaching surveillance (Simón *et al.*, 2012, 2013). A total of 23,403 ha (7,881 ha in Guadalmellato and 15,522 ha in Guarrizas) are under collaborative agreement. During the first year of releases, six (3 males:3 females; two breeding-aged couples) and five (2 males:3 females; one breeding-aged couple) individuals were soft-released in Guadalmellato and Guarrizas respectively. Soft-release enclosures are 4 ha pens that were built in areas of high rabbit density. Moreover, supplementary feeding is performed while lynxes stay inside the soft-release enclosures. From the second year onwards, both soft- and hard-releases were performed.

Post-release monitoring: Released individuals are monitored through telemetry (both VHF and GPS-GSM collars) and photo-trapping (Simón *et al.*, 2013). In soft-releases, individuals are observed 8 hours/day in order to evaluate their behaviour and potential interaction between individuals. When soft-release was performed inside the home range of a settled individual, fights across the mess between resident and released individuals were not uncommon. Once this problem was detected, no other soft-releases were performed inside settled adult territories. Reproduction events and productivity are identified by means of photo-trapping (Simón *et al.*, 2013). Mortality was mostly detected by means of telemetry. Post-release development has been similar in both between hard- and soft-released individuals, and between wild-caught and captive-bred ones. None of the individuals had to be recaptured due to a lack of adaptation to the environment.

In Guadalmellato, reproduction was confirmed since the first year of releases. One breeding female raised two cubs during the first year, three breeding females raised seven cubs during the second one, and four breeding females raised six cubs during the third one. Nine out of the 19 lynxes released in Guadalmellato were found dead within one year after the release. Of them, four individuals died (run over) in car accidents, three poached and two died due to unknown causes. Because of these results, conservation actions in Guadalmellato are currently focused on decreasing both road accidents and poaching.

In Guarrizas, no breeding was recorded in the first year, whereas two breeding females raised eight cubs during the second one. Five of the fifteen lynxes released were found dead within the first year after the release. Of them, two individuals were poached, one died in a car accident, another one as a result of a disease and the last one due to a fight. Conservation efforts in Guarrizas are being mainly directed to prevent poaching.

Release of Iberian lynx © Guillermo Lopez

Overall mortality during the first year after release was 41.2% (47.4% in Guadalmellato and 33.3% in Guarrizas). Although sample size is still low for solid conclusions, mortality during the first year was barely lower in wild-caught individuals than in captive-born ones (29.4% vs. 52.9%). Apparently, there were not differences in survival, behaviour or settlement between soft- and hard-released individuals. In the following years, releases of 5 - 10 genetically-selected individuals per area will be performed annually in order to achieve a rapid population increase. Moreover, re-introduction in a third optimal area will begin in 2014.

Major difficulties faced

- Measures to decrease fragmentation caused by infrastructures and its related risks are very expensive.
- The use of illegal non-selective hunting methods is frequent and its total eradication sometimes becomes a difficult issue.
- Prior to re-introduction, enough funding to perform the whole program must be ensured.

Major lessons learned

- Performing Iberian lynx re-introduction in a large area (>10,000 ha) of high rabbit density guarantees both settlement and reproduction of released lynxes.
- Soft-releases should not be performed inside the home range of a resident Iberian lynx.
- Both soft- and hard-releases can be successfully used in the Iberian lynx re-introduction.
- Risk due to both poaching and car accidents might be carefully considered and reduced before beginning an Iberian lynx re-introduction program.
- The support of the local population is essential to the success of an Iberian lynx re-introduction program.

Success of project

Highly Successful	Successful	Partially Successful	Failure
√			

Reason for success/failure:

- Steps are being made according to approved re-introduction program.
- Settlement and productivity are over the previsions in both areas.
- All released individuals have adapted well to the environment.
- Connection between Andújar-Cardeña and both re-introduction areas has been demonstrated by the movements of four different individuals.
- The social support to the re-introduction program is high, and the involvement of the population in the program increases every year.

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Re-introduction of the brown bear in the central Alps, Trentino, Italy

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Introduction

The brown bear (*Ursus arctos* Linnaeus, 1758) population of northern Italy is one of the smallest in Europe and is central for the restoration of the species in the Alps (Swenson *et al.*, 2000). The decline of the brown bear population in the Alps started during the 18th century due to human persecution and habitat loss and fragmentation (Mustoni *et al.*, 2003). Bears in northern Italy became isolated from the larger Dinaric-Balkanian population, and by the end of the 1990s, only few relict individuals survived in the central Alps, in the Trento province (Trentino) and the population was considered biologically extinct. In 1999 - 2002, nine bears from Slovenia were released in Trentino as part of a translocation program (Life-Ursus Project). Since then, the population has grown to 43 - 48 bears in 2012 (estimates are over 50 bears for 2013), and has expanded into part of the former range (Groff *et al.*, 2013). The modern Alpine ecosystem, a mosaic of natural and human dominated environments, poses the main challenge for the management and conservation of this population. Brown bears are globally listed as of Least Concern (IUCN), the population of northern Italy is in Annex II of the CITES (Washington, 1973) and its protection is regulated at the national and European level.

Goals

- Goal 1: Use translocation of brown bears from Slovenia to avoid the extinction of one of the large carnivores of the Alps, assure the continuity of brown bear presence in the region and preserve the legacy of the native brown bear population, re-establish a minimum viable population of 40 - 60 bears in the central

Brown bear © C. Frapporti


Typical bear habitat © Archives Forest Service

Alps in 20 - 40 years and a brown bear meta-population in the Alps in the longer term.

- Goal 2: Promote the co-existence between humans and bears by increasing awareness of the human population towards brown bears and the re-introduction project, through environmental education and media information, and involving local stakeholders in the project.
- Goal 3: Mitigate human-bear conflicts through protocols for damage evaluation, establishment of damage compensation schemes, prevention of damage to properties, and management of problem bears and of emergency situations.
- Goal 4: Monitoring and scientific research to measure success of the re-introduction and allow timely intervention if deemed necessary.
- Goal 5: Establish a network, at the national and international level, between the different relevant authorities to promote population level management.

Success Indicators

- Indicator 1: Number of founders surviving and reproducing after translocation, positive population growth and reproduction.
- Indicator 2: Preservation of the genetic legacy of the last bear population of the Italian Alps and maintenance of genetic diversity.
- Indicator 3: Habitat use and distribution of the population, connectivity with other bear populations in the eastern Alps.
- Indicator 4: Support to the project (number of local administrations, stakeholders and other associations adhering to the project, attitude of the public opinion during and after the translocation).
- Indicator 5: Level of knowledge acquired through monitoring, number and impact of research projects and activities.

Project Summary

Feasibility: A feasibility study was carried out by the former National Wildlife Institute (now ISPRA) to evaluate the environmental, organizational, administrative, socio-economic and normative aspects of a brown bear re-introduction in the central Alps in Italy based on the analysis of ecological, social and economical data (Dupré *et al.*, 1998). The study estimated that only 2 - 3 relict bears remained in Trentino, based on genetic analysis of feces collected in the region during the years preceding the project. Based on a sample area of 6,495 km^2, the study verified the existence of a minimum suitable habitat of

~1,700 km^2 for supporting a MVP of 35 - 50 bears, taking into account bear ecological requirements, environmental features, and human presence. The attitude of the human population living in non-urban areas was surveyed and found to be mostly (>70%) favorable to the re-introduction, despite lower levels of acceptance in some areas. The study also highlighted the necessity of improving prevention and compensation measures for damages possibly caused by bears. Finally, it was determined that, to achieve project objectives 9 bears (3 males and 6 females and approximately 2 - 6 years old) should be released in the area were the last relict bears of Trentino still existed. The Slovenian population should be the source of the translocated bears given the short time the populations have been separated, the behavioral characteristics, and the sustainability of the removal (stock taken from the Slovenian hunting quota). The study concluded that a re-introduction was feasible and could lead in the mid- to long-term to the successful re-establishment of the species in the central Alps.

Implementation: The main agencies responsible for the implementation of the project were the Parco Naturale Adamello Brenta, which first promoted the re-introduction, the Provincia Autonoma di Trento, which coordinates management activities, and the former National Wildlife Institute, which provides scientific support. Formal technical and administrative agreements were established with other local administrations, including neighboring provinces and countries where the bears were likely to expand.

The support of local stakeholders to the project was ensured through the involvement of the Hunting Association of the Trento Province, Trento WWF, and organizations of categories particularly affected by bear presence such as livestock farmers and beekeepers. Specific Guidelines were produced to define operational programs (i.e. monitoring, management of problem bears, damages and emergency situations, training of personnel, communication) and the specific roles of the various agencies involved. The Trento Province secured a budget for compensating damage losses and other management activities.

Translocations took place during four years, between 1999 and 2002. Bears were captured within two hunting reserves in southern Slovenia and released in the Parco Naturale Adamello Brenta in the western part of Trentino. An additional female was released to replace one that died in an avalanche shortly after release.

Brown bear with radio-collar © C. Groff

Post-release monitoring: All re-introduced bears were equipped with a VHF collar and two ear tags to allow precise determination of their position, at least twice per day, and evaluate potential risks to people and properties, therefore preventing situations of possible conflicts with humans. Radio-tracking was the main monitoring method from 1999 to 2003 and provided important data on survival, habitat use, and distribution of the translocated bears (Zibordi *et al.*, 2010). Radio-tracking through GPS/VHF technology is still used for close monitoring of problem bears. Starting in 2003, genetic monitoring became the principal mean to obtain demographic, reproductive, ecological, distribution and genetic information on the released bears and their descendants. The method is based on the analysis of the DNA extracted from biological samples, mostly bear hair and feces collected non-invasively in the field, using a variety of sampling techniques, but occasionally also tissue, blood, and bones retrieved during capture operations or from bear carcasses (De Barba *et al.*, 2010; Groff *et al.*, 2013). Data from radio and genetic monitoring is complemented with additional information from traditional sign survey, visual observation (i.e. female with cubs), and camera traps.

Most released bears survived and reproduced in Trentino (7 of the 9 founders, 2 males, 5 females); since the translocation the population grew rapidly to estimated 43 - 48 bears in 2012 (the threshold of 50 individuals will likely be met in 2013) due to high reproductive rate (34 documented birth events, at least 69 cubs in 2002 - 2012); survival rates over 11 years were 81,8%, 92,9%, 91,3% for cubs, juveniles, and adults respectively (Groff *et al.*, 2013); and the released bears and their descendants started to recolonize the former bear range in the Alps. Since the beginning of the project, 14 bears were found dead, additional 13 bears have not been detected through genetic monitoring for at least the two past years and two bears were placed in captivity and two bears dispersed outside the study area (Groff *et al.*, 2013).

The population is still demographically isolated; however long distance male-biased dispersal, from Trentino to the east and from Slovenia to the west, has recently resulted in the partial overlap, without gene flow, of the two bear populations. As a consequence, initially high genetic diversity is declining, five inbred litters, out of a total of 30 litters have been detected through pedigree reconstruction, and the effective population size (Ne) remains small (De Barba *et al.*, 2010). Repeated opinion surveys showed a dramatic decrease in the public support, despite the communication campaigns, and the efforts for damage prevention and compensation.

Major difficulties faced

- Higher than expected conflicts with humans and management of problem bears.
- Human caused mortality especially in neighboring countries.
- Lack of efficacy and coordination for the management and potential removal of the few problem bears, and consequent dramatic decrease in public support.
- Establishment of gene flow with the Slovenian population not yet recorded.
- Need for effective trans-national agreements for bear management in the Alps.

Major lessons learned

- Importance of support and involvement of local population and stakeholders, and of agreements with administrations affected by bear presence.
- Importance of long term, science based post-release monitoring and research.
- Importance of increasing awareness and education of human population and delivery of project status and results.
- Importance of trained scientists and field teams.
- Importance of effective and prompt measures to manage problem bears.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- Survival and reproduction of majority (77%) of founders; high reproductive rates of re-introduced population and achievement of the minimum demographic objective of >50 bears in less than 20 years.
- High initial levels of genetic diversity comparable to the source Slovenian population.
- Geographic expansion and beginning of recolonization of former bear habitat in northern Italy and neighboring countries (Switzerland, Germany and Austria).
- Inter-regional agreements ensuring large scale monitoring, damage compensation and prevention, management of problem bears and emergencies, personnel training, communication to human population. In this context, an Alpine action plan for the Conservation of the Brown Bear (PACOBACE), endorsed by the Italian Ministry of Environment, ISPRA, and relevant regional administrations, was produced in 2010 for the Italian Alps. Efforts are presently carried out to officially recognize the international alpine bear group already operating since 2006.
- Effective and adaptive management strategies (monitoring, damages and emergencies, personnel training, communication).

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Translocation of Hawaiian monk seals in the Hawaiian Archipelago and Johnston Atoll, USA

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Introduction

The Hawaiian monk seal (*Monachus schauinslandi*) is one of the world's most endangered marine mammals, numbering approximately 1,200 individuals and decreasing at a rate of about 3% per year. Hawaiian monk seals occur throughout the 2,600 km-long Hawaiian Archipelago, which consists of two regions: the main Hawaiian Islands (MHI, with eight primary high islands) and the Northwestern Hawaiian Islands (NWHI, made up of small coral islands, low-lying atolls, and steep basalt islands). Most monk seals reside in the remote NWHI, and a small population occurs in the MHI. There are rare and sporadic reports of seals visiting Johnston Atoll, approximately 800 km south of the Hawaiian Archipelago. Adult monk seals are approximately 220 cm in length and can weigh over 200 kg. They give birth, nurse, rest and molt on land and forage on a wide variety of prey on the sea floor, sometimes at depths exceeding 500 m. Identified threats include food limitation, shark predation, conspecific male aggression, entanglement in derelict marine debris, fishery interactions, and intentional killing by humans. The species is Critically Endangered under the IUCN, endangered under the U.S. Endangered Species Act, depleted under the U.S. Marine Mammal Protection Act, and listed on CITES Appendix I.

Goals

- Goal 1: Translocation to alleviate immediate risks of mortality, such as shark predation or conspecific male aggression.
- Goal 2: Translocation from area of lower to higher long-term survival probability (e.g., due to more favorable foraging conditions).

Hawaiian monk seal © Jon Brack

- Goal 3: Translocate aggressive male seals to mitigate injury and mortality of conspecifics.
- Goal 4: Translocate seals in human-populated areas to mitigate undesirable or dangerous human-seal interactions.

Success Indicators

- Indicator 1: Successful execution of translocations (capture, transport and release).
- Indicator 2: Acceptably low rate and distance of dispersal from release area.
- Indicator 3: Post-release survival effects are acceptable. That is, survival rate of translocated seals matches that of comparable individuals at the release site. Or, (depending on the goal of translocation) survival of translocated seals is improved relative to survival at the capture site.
- Indicator 4: Post-release foraging behavior and habitat use of translocated seals is similar to comparable seals at the release location.
- Indicator 5: Intended goals of translocations were achieved.

Project Summary

Feasibility: The Northwestern Hawaiian Islands (NWHI) and Main Hawaiian Islands (MHI) differ in nearly every aspect relevant to monk seal conservation (Baker *et al.*, 2011). The remote NWHI are part of the Papahānaumokuākea Marine National Monument, a vast marine-protected area where coral reefs and associated fish populations are considered robust and fishing and other *in situ* human impacts have been minimized. In contrast, the MHI are characterized by a large human population and nearshore marine ecosystems severely impacted by physical alteration, heavy fishing pressure, and pollution. Nevertheless, monk seals appear to be thriving in the MHI (Baker *et al.*, 2011), while the NWHI subpopulations in aggregate are declining, believed largely as a result of food limitation leading to low juvenile survival. MHI seals may enjoy relatively low intra-specific competition (because the number of seals is still small) and low inter-specific competition (because large predatory fish competitors have been greatly reduced by fishing). Johnston Atoll is very isolated and encloses four small, low-lying islets. It is considered part of the monk seal's range because seals naturally occur there, though only rarely and typically only singly.

In the NWHI, strict quarantine protocols minimize the risk of introducing invasive species to these fragile island ecosystems characterized by a high degree of endemism. At some sites, access is controlled to protect cultural resources and archaeological remains. In the MHI, social factors play a large role in monk seal translocations. Public attitudes toward monk seals are diverse. Undesirable seal interactions with people have motivated several seal relocations, and sensitivity to public sentiment is an important element of translocation decision-making.

Implementation: Translocation has been a tool for Hawaiian monk seal conservation for the past 30 years. A total of 259 seals were translocated during 1984 - 2009 (Baker *et al.*, 2011; Norris, 2013). Seals were transported from just a few kilometers within an island or atoll to over 2,000 km between subpopulations. Consequently, the cost and complexity of associated logistics varied greatly with

Translocating a weaned monk seal pup from area of high shark predation © Monica Bond

the scale of translocations. The NWHI are primarily accessible only by large seagoing vessels. The exception is Midway Atoll (previously also Kure Atoll and French Frigate Shoals), which aircraft may access. Within NWHI atolls, seals have been transported aboard small boats or carried by hand. In contrast, the MHI shorelines are largely accessible by some combination of aircraft, vessel and automobile, and these have all been used for translocations within the MHI. Translocations between the MHI, NWHI and Johnston Atoll were accomplished using large vessels or aircraft.

Attention to disease transmission risk has varied over time and with the nature of translocations. Prior to the 2000s, no testing for disease exposure was conducted. Thereafter, potential variation in disease exposure among subpopulations has been assessed, and when seals were translocated among subpopulations, individuals were subject to health screening (Norris, 2013). Further, Schultz *et al.* (2011) found that the Hawaiian monk seal is comprised of a single panmictic population, so that there are no concerns regarding genetic consequences of translocations. Most seals had “hard releases” on land, in that they were simply let go on the beach. In 1990 - 1991, six weaned pups instead had a “soft release”, as they were held for 1 - 2 months in shoreline pens at Kure Atoll and offered live fish prior to release. For logistical reasons, 21 adult male seals were released from a ship in nearshore waters of the MHI in 1994 and 12 weaned pups were released from a small boat within 100 m of shore at Nihoa Island in 2008 - 2009.

Post-release monitoring: Most seals are individually identifiable by applied tags, temporary pelage bleach marks, and photographic identification using natural markings. Post-release monitoring and program assessment largely relied on resighting translocated individuals over time and comparing their movement and survival to appropriate “control” seals. The following was reported by Baker *et al.* (2011) and Norris (2013). Recently weaned pups (with little or no at-sea foraging experience) exhibited high fidelity to release sites commensurate with that shown by untranslocated pups to their birth location. In contrast, juvenile and adult seals tended to stray from their release locations farther and sooner. Nevertheless, when 21 adult male seals were moved over 1,000 km from Laysan

Island (NWHI) to the MHI, they subsequently dispersed among the MHI; however, only one was observed to return to the NWHI. Translocated seals' survival rates were indistinguishable from those of comparable seals native to the release sites. Further, where comparisons could be made, seals translocated to improve their survival appeared to fare better than comparable seals remaining at their natal locations.

Detailed post-release telemetry tracking was conducted on 12 pups translocated from French Frigate Shoals to Nihoa Island (Norris, 2013). Similar post-release movement patterns, diving activity, and habitat use were observed for translocated and non-translocated monk seal pups at Nihoa Island and other sites in the Hawaiian Archipelago, indicating monk seal pups had normal foraging activity following translocation.

Major difficulties faced

- Once seals had become habituated to people, translocations within the MHI to mitigate their interactions with people typically failed. Habituated seals usually dispersed from relatively remote release sites (often repeatedly when relocated and released multiple times) and continued to seek out human contact. These cases tended to ultimately result in the seals being taken out of the MHI (to Johnston Atoll, the NWHI or permanent captivity) to address public and seal safety concerns.
- Adult males and a subadult male released at Johnston Atoll apparently did not persist there long post-release. Some may have died there, whereas others (fitted with satellite tracking devices) departed the atoll soon after release and were never resighted.
- Small sample sizes sometimes inhibited robust statistical inference.
- Post-release monitoring effort for many of the translocations conducted prior to the late 1990s was inadequate.
- Budgetary and logistic constraints limited post-release visual monitoring of the most recent (2008 - 2009) translocations to Nihoa Island. Available information suggested these translocated pups likely fared considerably better than those at their natal site, but perhaps not as well as native Nihoa Island pups. Imprecise survival estimates due to low monitoring effort hampered project evaluation.

Major lessons learned

- There is little risk to Hawaiian monk seals associated with the mechanics of capture, transport, and release. Of 259 seals translocated, only 3 (1.2%) died during translocation procedures, including 2 adult males and 1 weaned pup. One of the adults died while being restrained, while the second adult and the pup died while being held in temporary captivity. Cause of death in all 3 cases could not be determined. Capture stress, pre-existing conditions or both may have been involved. A wide variety of transportation methods may be safely employed, including carrying seals on foot, transporting in small boats, large ships and aboard aircraft. Whenever feasible, releasing translocated monk seals on land is preferred to a boat-based release, especially for young seals.

Hawaiian monk seal pup on research vessel with GPS telemetry instrument © Hung Tran

- Weaned pups are most amenable to translocation. They are robust to handling and transport, show high fidelity to release sites, and apparently survive as well as comparable native pups at the release location. Older seals also appear to exhibit favorable survival rates post-release (with the exception of those taken to Johnston Atoll) but tend to disperse sooner and more widely.
- Based on existing information, Johnston Atoll is not a viable release site for monk seals.
- Most intended goals of the translocations were consistently achieved. Notably, undesirable human-seal interactions can be successfully prevented through translocation of young seals (weaned pups) prior *to* habituation to humans. However, once seals have become habituated to people, translocation within the MHI is unlikely to resolve problem interactions.
- The 1994 translocation of 21 adult males to the MHI convincingly achieved the desired goal of reducing female seal mortality at Laysan Island (NWHI) (Johanos *et al.*, 2010). At that time, monk seals existed in low numbers and had rarely been seen by Hawaii residents. Scientists and managers failed to involve the public in decision-making, nor was there follow-up public education. Apparently as a result, some Hawaii residents believe that monk seals are not native to the MHI and do not belong there. Thus, while the immediate goal of the translocation project was achieved, this action also contributed to persistent animosity towards seals and a lack of support among some members of the public for monk seal conservation, even two decades hence.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- Monk seals are clearly robust animals and can readily withstand temporary handling and captivity. We believe the rarity of translocation-related mortality has also resulted from strict adherence to cautious handling and transport protocols.

- In the short term, pups undergo a post-weaning fast and remain relatively sedentary and mostly on shore for 1 to 2 months after weaning. Pups translocated during this period tend to stay put where released and slowly expand their range once they begin to forage. This behavior facilitates post-release monitoring and provides a measure of confidence that pups will not rapidly disperse from the habitat selected for release to perhaps less desirable habitat.
- In the long term, most monk seals translocated at all ages remained in the general region where they were released and did not return to their natal areas.
- “Hard” releases with no acclimation period work well for this species. Pups were typically released immediately on shore and older animals either on or near shore. This meant captive time was limited to that required for health screening and transport (which typically can be done simultaneously). Consequently, cost of captive care, risk of capture-related health complications and potential for human habituation were minimized.
- The long-term, detailed monk seal demographic database and the fact that most seals are individually identifiable both improve design of translocation actions and facilitate post-release monitoring.

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Re-introduction and supplementation of large antelopes and zebra to Debshan Ranches, Central Zimbabwe

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Introduction

As their habitats are reduced and fragmented by the expansion of agriculture and human settlement, numbers of many African antelopes are declining (East, 1999). In southern Africa, the economic value of wildlife, often realised through trophy hunting, has promoted conservation on private land. During 1991, Debshan Ranches initiated a project to boost small populations of wild herbivores and to re-introduce species that historically occurred there. Common eland (*Tragelaphus oryx*), wildebeest (*Connochaetes taurinus*), giraffe (*Giraffa camelopardalis*) and waterbuck (*Kobus ellipsiprymnus*) were re-introduced, and sable antelope (*Hippotragus niger*), bushbuck (*Tragelaphus scriptus*) and Burchell's zebra (*Equus quagga*) were released to supplement existing populations. The conservation status of all these species is Least Concern (IUCN Red List). Shangani Ranch (c. 480 km^2) and De Beers Ranch (c. 200 km^2) lie in central Zimbabwe. Mean annual rainfall is *c*.600 mm, with a single rainy season during November - March (Dunham *et al*., 2003). The main vegetation types are: *Terminalia sericea* woodland and wooded grassland; *Brachystegia–Julbernardia* woodland; *Colophospermum mopane* woodland and shrubland; *Acacia-Combretum* woodland on alluvial soils; and hydromorphic grassland. The ranches are used primarily for beef production, but support various wild herbivores and predators (leopard, cheetah and black-backed jackal, and occasionally lion and spotted and brown hyenas).

Zebra and cattle share range on Debshan Ranch
© Susan Swanepoel

Goals

- Goal 1: Create self-sustaining populations

Hard release of bushbuck from crate at Debshan ranches in 2011 © Susan Swanepoel

of the re-introduced species on the ranch(es) where those species were released.

- Goal 2: Create self-sustaining populations of the supplemented species on the ranch(es) where those species were released, with the post-release populations being more numerous than the pre-release populations of the same species.
- Goal 3: Increase the genetic diversity of the previously-small populations of supplemented species.

Success Indicators

- Indicator 1: Sightings of released animals in the general vicinity of release sites (indicating that freed animals had both survived and not left the ranch where they were released).
- Indicator 2: Sightings of young animals (indicating that released animals had bred successfully).
- Indicator 3: Long-term increases in the population numbers on the ranches.

Project Summary

Feasibility: The ranches were established during the early 20th century and records from early European travellers (e.g. Baines, Selous) and ranch staff showed that the species earmarked for re-introduction previously occurred there. The ranches still contained significant populations of other wild herbivores such as impala, tsessebe, kudu and warthog. Hence, it was believed that the vegetation was suitable for the re-introduced species - most likely the original populations were eliminated by excessive offtakes (e.g. to reduce competition with cattle), not by habitat changes. For the species earmarked for supplementation, it was thought that the populations were kept small by Allee effects (possibly predator-driven), not by issues of habitat suitability. Sable antelope, waterbuck, wildebeest and zebra are grazers, giraffe and bushbuck are browsers, and the eland is a mixed feeder.

Implementation: All the released animals were purchased through private sale or game auctions, from ranches in Zimbabwe. For all the species, there is only a single subspecies within Zimbabwe (Lorenzen *et al.*, 2012) and so the main concerns about moving animals around the country were veterinary ones. What was available for purchase placed some limitations on the size and age/sex composition of the groups released. Prices varied between species and high prices limited the numbers of the rarer species that could be released. National regulations for controlling foot-and-mouth disease required that the animals came from areas where both the disease and potential carriers (e.g. buffalo) were absent (the 'green zone'); or, for those captured elsewhere, that they were tested for foot-and-mouth disease by the veterinary authorities after capture. The animals came from southern Zimbabwe, the Midlands, or Doma in north-east Zimbabwe. Except for animals from auction sales, most were transported to their release site immediately after being caught by professional game capture teams, usually during the cooler dry-season months. Bushbuck were individually crated during transport, but other species travelled in groups. At the release site, the animals were freed into a *boma* (pre-release pen) with high (approximately 2 m) sides of black, opaque plastic sheeting, and measuring c. 50 m x 50 m. The initial groups were kept in the *boma* for up to 14 days, during which they were given food (commercial game pellets and cotton seed) and drinking water. They were freed by removing part of the boma side and letting them find their own way out. After the apparent success of the early releases, and given the difficulty of confining large antelopes in even high-sided bomas (some jumped the sides), later release groups were confined to the *boma* for just one night, to prompt group cohesion, before release. Later still, some less financially-valuable animals (zebra) or solitary species (bushbuck) were freed immediately on arrival at the ranch ("free-released").

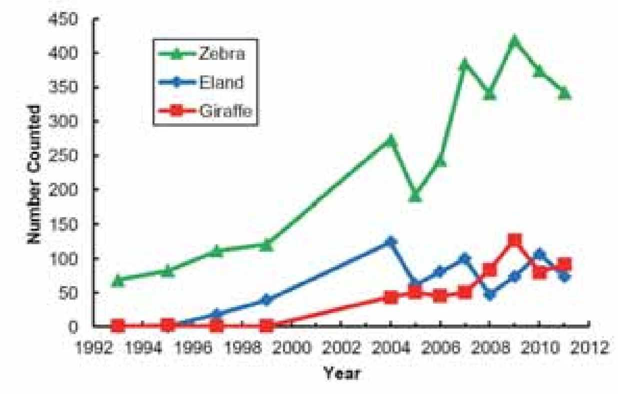
More valuable species (waterbuck and sable antelope) were released in Shangangwe game-fenced enclosure of c. 20 km^2 on De Beers ranch. Although it contained cattle, this enclosure was surrounded by a 2 m, 10-stand, electric fence that served to prevent the wild herbivores leaving the ranch, and to facilitate their

Table 1. The total numbers of each species released on the two ranches, the range of group sizes and the years when the releases occurred (R = Re-introduction; S = Supplementation)

Species	Release Ranch				Number in release group	Years of releases
	Shangani		De Beers			
Zebra	101	S	89	S	8 - 25	1991 - 1997
Eland	94	R	102	R	10 - 41	1991 - 1997
Wildebeest	0		30 *	R	7 - 23	1998 - 1999
Giraffe	6	R	0		6	1997
Sable antelope	5	S	28 *	R	5 - 18	1993 - 1998
Waterbuck	0		43 *	R	2 - 15	1995 - 1998
Bushbuck	0		5	S	-	1997
	58	S	24 *	S		2011

* Released in the Shangangwe enclosure of c.20 km^2

Fig. 1. The numbers of zebra, eland and giraffe counted during total-area counts from a helicopter of the wildlife on Shangani Ranch, Zimbabwe.



monitoring. Wildebeest were also released here in order to contain them, because wildebeest may carry a virus that causes malignant catarrhal fever, which is potentially fatal to cattle.

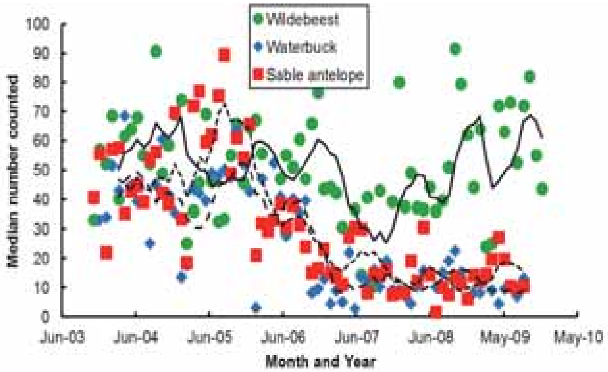
Post-release monitoring: There were three forms of post-release monitoring. First, there were incidental observations of groups of re-introduced or supplemented species during cattle management operations. Sightings provided information about the spatial distribution and dispersal of released animals, and the presence of juveniles in the groups indicated successful breeding. Secondly, whenever cattle management required cattle in a paddock to be rounded up, the staff - working on foot - searched the entire paddock and recorded the numbers of all wild herbivores that they saw. In addition to providing information on spatial distribution and breeding, these data also provided indices of abundance. Thirdly, usually every two years, all large, wild animals on each ranch were counted during a total-area survey, conducted from a helicopter flying at low level along parallel flight-lines.

The helicopter surveys revealed that eland and zebra populations became established on Shangani Ranch (Figure 1). A giraffe population also became established, even though just six animals (all adult females) were released. The increased number of giraffe clearly resulted at least partly from immigration, either because immigrants were attracted by the released animals, or because of disruption caused on neighbouring properties by Zimbabwe's agrarian reform program or both. Giraffe releases were discontinued because monitoring revealed that the population was increasing as a result of this immigration. Foot patrols revealed that the releases of waterbuck and sable antelope in the

Eland in plastic walled boma © Kevin Dunham

Shangangwe enclosure were initially successful, with numbers peaking during 2005, approximately seven years after the last releases (Figure 2). But during 2006, the number of both species declined to about one-third or less of their former level and then remained generally low. Wildebeest numbers appeared to decline later, during 2007, but then increased.

Fig. 2. The median numbers of wildebeest, waterbuck and sable antelope counted monthly during foot patrols of Shangangwe enclosure on De Beers Ranch, Zimbabwe. Temporal trends shown by lines indicating the 5-month running means (solid line = wildebeest; small dash = sable; large dash = waterbuck).



Major difficulties faced

- The monitoring schemes were good at revealing population trends, but did not reveal the reasons for any population changes. It is still unknown why the sable antelope and waterbuck populations in the Shangangwe enclosure declined so dramatically during 2006: possibly some animals were poached, or chased out of the enclosure. But the absence of an increase in either population during the following three years suggests that both were now being regulated, possibly in some density-dependent fashion, or as a result of a new or additional mortality factor.
- The ranches are large, but home ranges of eland are often also large. The fate of animals that left the ranches was often unknown (one released group of eland reappeared on Shangani Ranch after an absence of two years).
- The groups of the less-common and thus more-costly species that were available were often small, less than the 10 - 20 individuals that was the preferred group size.
- Roan antelope (*Hippotragus equinus*) was a species that would have been re-introduced if groups for release could have been purchased at a reasonable price.

Major lessons learned

- Need to release a sufficient number (suggest 50 - 100) of animals of target species, including males and females of a range of ages.
- Hard releases (freeing the animals immediately they arrived at the release ranch) seemed to be as effective as soft releases (when animals spent up to two weeks in a pre-release boma).

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- Funds were available to obtain and release significant numbers of animals over a period of several years.
- The ranches were large, but nonetheless some released animals dispersed off the ranches. In the early stages, the ranches had a 'soft edge' because at least some of the neighbours were friendly towards wildlife. More recently, Zimbabwe's agricultural reform programme has hardened the edges. This resulted, at least initially, in some species (e.g. giraffe, elephant) finding refuge on the ranches. But in the long-term, the presence of wildlife on the ranches has attracted poachers who kill wildlife for bushmeat or trophies (e.g. elephant tusks, zebra skins). The high demand for meat in neighbouring mining communities has promoted commercial poaching for bushmeat.
- Financial returns from trophy hunting on the ranches were primarily used to fund the anti-poaching activities necessary to maintain the wildlife populations.

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Re-introduction of Père David's deer "Milu" to Beijing, Dafeng & Shishou, China

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Introduction

Milu (*Elaphurus davidianus*) is listed as "EW" in IUCN Redlist, and is listed as National Protected Wild Animal in China. Milu was extinct in wild in China. A captive herd was assembled at Woburn Abbey, UK around the turning 20th century. The Duke of Bedford donated 36 Milu to the Beijing Milu Park in 1985 and 1987. Another group of 39 deer from five British zoos was re-introduced to Dafeng Milu Reserve on coastal marsh site in Jiangsu Province. Milu population in Beijing Milu Park increased from 39 to 200 from 1985 - 2013, during the period over 300 Milu were relocated to more than 50 reserves and parks all over the country while Milu in Dafeng reserve increased to nearly 2000 in 2013. Since 1997, deliberate releases to the wild have taken place in Dafeng (Hu and Jiang, 2002). Ninety-one Milu were translocated from Beijing Milu Park to Tianezhou, Shishou, Hubei in 1993 and1995, where the Shishou Milu Reserve was established. A flooding in Yangtze River in 1998 resulted in several cohorts of Milu leaving the initial release area and forming permanent herds in other parts of the province, as well as around Dongting Lake in Hunan Province (Maddison *et al.*, 2012).

Goals

- Goal 1: Identification of potential re-introduction sites within historic range of Milu.
- Goal 2: Creating nature reserve and parklands for hosting the re-introduced Milu.
- Goal 3: Forming self-sustained re-introduced Milu population and gradually using the re-introduced Milu herd as source for further relocation.
- Goal 4: Monitoring of disease and parasites in the re-introduced Milu populations and managing habitat at re-introduction sites.

Milu in Dafeng Reserve: 2010 © Jiang Zhigang

- Goal 5: Releasing Milu to wild and re-establish wild Milu populations in the country.

Success Indicators

- Indicator 1: Establishing healthy self-sustaining breeding stocks of re-introduced Milu.
- Indicator 2: Relocation of the re-introduced Milu to other suitable sites in its historical range.
- Indicator 3: Establishing wild populations of Milu in its historical range.
- Indicator 4: Using Milu as conservation education model to promote conservation consciousness.
- Indicator 5: Learning experience from the case of Milu re-introduction for re-introducing other wild extinct species such as Przewalski's wild horse (*Equus przewalskii*), Saiga (*Saiga tatarica*) and one-horned rhino (*Rhinoceros unicornis*) in the country.

Project Summary

Feasibility: After last glacial period, Milu was restricted to swamp and wetland in the region south of 43°N and east of 110°E in China. Population of Milu declined due to human hunting and land reclamation as human population expanded in Holocene. Milu was finally extinct in the field (Cao *et al.*, 1992). Nanyuan Royal Hunting Garden in the Qing Dynasty (1616 - 1911) hosted a last herd of Milu, the landscape in the 200 km^2 hunting garden in south suburb of Beijing was a predominantly a wetland of swamp, ponds and lakes during that period. At the end of 19th century, wall of the garden was first destroyed by a heavy flood in the Yongding River, and then by the cannon fire of the allied foreign forces during the Second Opium War, the Père David's deer escaped and were hunted.

Before the demise of the royal herd of Milu in the Nanyuan, Milu had been introduced into Europe. During the last decade of the 19th century, the 11th Duke of Bedford gathered all last 18 Milu in the world to form a breeding herd at the Woburn Abbey. The heavily inbred Milu safely passed though the genetic bottleneck of inbreeding and adopted the vast open parkland of mid-England estate. In the 1950s, number of Milu reached several hundreds (Beck & Wemmer, 1983). After culture revolution, the feasibility of re-introducing Milu to China was explored.

Implementation: The first conservation re-introduction of Milu included two groups of 20 (5 males:15 females) and 18 (all females) in 1985 and 1987, respectively. After a careful search and evaluation by a group of zoologists, botanists, wildlife managers and officers the relic site of the Nanyuan Royal Hunting Garden was chosen as the site of re-introduction. For the re-introduction, the Beijing Milu Park (39°07'N, 116°03'E) was established.

Beijing Milu Park is 60 ha in area with Annual average temperature of 13.1℃, and average precipitation of 600 mm. The land was dominated by reed (*Phragmites australis*), and grasses, such as *Eleusine indica*, *Ersagrostis cilianensis*, *Digitaria* *sangunalis* and *Setaria viridis*. Where the grass is overgrazed, *Amaranthus roxburghianus* dominates the vegetation. Since the re-introduction, Milu over grazed on natural vegetation in summer and autumn, thus original vegetation inside the park was damaged by over grazing and droughts. The park managers started to plant artificial grasslands and rebuilt wetlands in the park. The deer in the park receive supplemental feeding year round (Jiang *et al.*, 2008).


Author in Dafeng Reserve in 2007

Further population growth in Beijing Milu Park was restricted by its limited size, thus, the park translocated its Milu to reserves and parks. More than 300 Milu were sent over 50 sites all over China. The most important one was relocation to Tianezhou on the riverside of Yangtze River in 1990s. Ninety-one Milu were translocated to Tianezhou, Shishou, Hubei Province in 1993 and1995, where the Shishou Milu Reserve was established. A flooding of the Yangtze River in 1998 resulted in several cohorts of Milu leaving the initial release area and forming permanent herds in other parts of the province, as well as around Dongting Lake in Hunan Province (Maddison *et al.*, 2012). The second re-introduction of Milu was carried out in August of 1986. A group of 39 Milu was selected from five UK zoos. An even more extensive survey in eastern China for potential re-introduction site was conducted. Finally, the Dafeng State Forestry Farm was chosen which is located on the coast of Yellow Sea and was lightly populated. Dafeng Milu Natural Reserve (33°05’N, 120°49’E) was established to host the re-introduced Milu..

The Dafeng Milu Natural Reserves is 2 - 4 m above sea level, with a sub-tropic monsoon type climate. Annual average temperature is 14.1℃ and average annual precipitation is 1,068 mm. The vegetation is dominated by cogongrass (*Imperata cylindrica*), reed (*Phragmites australis*), locust false-indigo (*Amopha fruticosa*) and locust (*Robinia pseudoacacia*) (Jiang *et al.*, 2008). The original size of the Dafeng Milu Natural Reserve was 10 km^2, with 3 fenced paddocks of 273 ha. The reserve purchased another 30 km^2 land in 1995. In 1997, the reserve was approved by the National Nature Reserve Commission as a national nature reserve (Jiang *et al.*, 2000b).

Post-release monitoring: Wardens and veterinarians of Beijing Milu Park, Dafeng national nature reserve and Shishou Milu national reserve routinely

Close-up of Milu

closely monitored the re-introduced Milu. Researcher and graduate students from the institute of Zoology, Chinese Academy of Sciences, Chinese Forestry Academy and universities conducted research projects including population monitoring on the introduced Milu in Beijing Milu Park, Dafeng national nature reserve and Shishou Milu national reserve. Three international workshops on management and research on the re-introduced Milu were held in Beijing Milu Park in 2006 and Dafeng reserve in 2011 and 2012, respectively. Recently, a team is monitoring the field released Milu in coast marsh of Dafeng with satellite collars. Many papers have been published in peer-reviewed journals.

Major difficulties faced

- Populations of Milu quickly reached the carry capacities of Beijing Milu Park, Dafeng Milu nature reserve and Shishou Milu nature reserve.
- Lack of further field releasing sites for Milu in the country.
- Disease and parasites may break out occasionally in Milu populations.
- Released Milu may damage crops and thus cause conflict of interests between reserves and local people.
- The problem of low genetic diversity may be still potential threat to the survival of Milu, though we do not notice major phonological change in the population.

Major lessons learned

- A thorough investigation of cause of field extinction and biology of the species is prerequisite for successful re-introduction.
- As first step of re-introduction, the Milu were released to fenced paddocks of a large size (100 ha in Dafeng) under close monitoring and supplemented with feed in winter.
- Actively looking for additional relocation sites such as parks, zoos, safaris and nature reserve as the number of re-introduced Milu increased.
- Expanding the size of nature reserve if possible, in case of Dafeng nature reserve, more coastal march lands were acquired for the field released deer in the reserve.
- A national level coordination scheme is needed for further field release and population genetic management.

Success of project

Highly Successful	Successful	Partially Successful	Failure
√			

Reason(s) for success/failure:

- Milu is perceived as a national conservation priority and a flagship species in the wetland ecosystems of the country.
- A consulting body for Milu re-introduction was formed, field surveys were conducted in the former range of Milu and a master plan for Milu re-introduction was drawn by national wildlife management authority.
- Local governments welcomed the implementation of re-introduction of Milu, because the Milu is legendary animal in Chinese history.
- Parks and natural reserves were established for the re-introduced Milu with veterinarians, wardens and budget from the local governments.
- Scientists conducted researches on ecology, behavioral, reproductive, and genetic as well as disease prevention in Milu.

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The re-introduction of Lichtenstein's hartebeest to Malilangwe Wildlife Reserve, south-eastern Zimbabwe

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Introduction

Lichtenstein's hartebeest (*Alcelaphus buselaphus* ssp. *Lichtensteinii*) is an African savanna antelope inhabiting the ecotone between woodland and seasonally flooded grassland (Booth, 1980). Historically the species occurred from Tanzania southwards through central Africa to north-eastern South Africa. Although vulnerable to poaching, the species is labeled "of Least Concern" in the IUCN Red List of Threatened Species because key populations in Tanzania and Zambia are currently stable. At the end of the 19th century the species was widely distributed in what is now Zimbabwe (Selous, 1893). However, by the 1960's numbers had declined dramatically, with only a few herds remaining; these being confined to a few privately owned ranches in the south-east (Booth, 1980). Lone Star Ranch harboured the largest population, with approximately 38 animals recorded in 1976 (Booth, 1980). However, by 1993 drought-induced mortality had reduced this population to one known animal (Colin Wenham, pers. obs.).
In 1994, The Malilangwe Trust purchased Lone Star and Maranatha Ranches to form Malilangwe Wildlife Reserve (see Clegg & O'Connor, 2012 for a biophysical description of the reserve). A principal objective of the Trust is to restore the historic biodiversity of the reserve, and consequently re-introduction of Lichtenstein's hartebeest became an important management goal.

Lichtenstein's hartebeest © Bradley Fouche

Goals

- Goal 1: Determine whether suitable habitat for Lichtenstein's hartebeest exists after the catastrophic droughts in 1983 and 1992.

- Goal 2: Establish a viable nucleus of breeding animals within a fenced enclosure.
- Goal 3: Establish a self-sustaining, free-ranging population by releasing animals from the breeding nucleus onto the reserve.

Success indicators

- Indicator 1: Number of animals in the fenced breeding enclosure is growing at or close to the maximum intrinsic rate of increase.
- Indicator 2: In the absence of catastrophic events, the established free ranging population is predicted to remain viable for 50 years.

Project Summary

Feasibility: On Lone Star, Booth (1980) showed that hartebeest selected for shallow, seasonally waterlogged, grassland depressions at the headward ends of drainage systems in dry woodland or bush vegetation. After Booth's study in 1976, these vegetation communities, which are known as dambos, underwent compositional and structural changes in response to severe droughts in 1983 and 1992. Consequently, in 1996 a study was conducted to determine whether suitable habitat for Lichtenstein's hartebeest still existed at Malilangwe (Clegg, 1999). Despite significant compositional changes to the herbaceous layer, and some bush encroachment by woody plants, the dambos at Malilangwe still provided suitable habitat, so it was decided to go ahead with the re-introduction.

Implementation: A breeding nucleus of 30 animals (1 adult male and 29 adult females), that had been sourced from Choma, Zambia (10 in 1996 and 20 in 1998) were kept in quarantine pens at Triangle for 21 days before being released into a 500 ha enclosure that had been constructed in prime hartebeest habitat at Malilangwe. By 2002, the number of animals had increased to 72, but mortality of 20 (18 from exposure) dropped the number to 61 in 2003. To spread the risk of further catastrophic mortality, a group of 6 animals was moved in 2004 to a second 500 ha enclosure that had been constructed in the north-east of the reserve. Animals in the enclosures were monitored daily by scouts on foot who recorded the age and sex of each individual and the cause of any mortality. The population in the first enclosure increased at an average rate of 30 % per annum, which is close to the maximum rate of increase for an antelope of this size. In June 2004, 24 animals (1 adult male, 12 adult females, 4 sub-adult males, 3 subadult female, and 4 juveniles) were released from the first enclosure; the remaining 26 being retained.

By October 2006, the free-ranging population had increased to an estimate of 51, with a captive population of 43 (33 in the first enclosure and 10 in the second). In 2007, the remaining 33 animals (3 adult males, 11 adult females, 4 subadult males, 6 subadult females, and 9 juveniles) in the first enclosure were released onto the reserve, and the enclosure dismantled. The captive animals in the second enclosure had increased to 12, and were retained as an insurance policy. This population has grown at an average rate of 17 % per annum, and only in the

Attaching a radio-collar on hartebeest

last few years has it entered the exponential phase of a logistic growth curve (the population stood at 26 in 2012).

Post-release monitoring: Since 1999, an annual census of the large mammal species has been conducted at Malilangwe using a helicopter and distance sampling techniques. In this way, estimates of the free-ranging hartebeest population have been derived annually from 2004 to 2012. After an initial increase from 24 animals in 2004 to 81 in 2007 (the population was boosted by a second release of 33 animals in 2007), the free-ranging population has shown a steady decline to an estimate of 60 animals in 2012. The main cause for this decline appears to be unsustainable levels of predation of adult females. Small populations are highly sensitive to loss of the adult female age class, and mortality of as few as 4 adult females per year can put a population of <100 into decline (Capon, 2011).

Although numbers increased rapidly in the enclosures, current levels of predation by lion and other large carnivores will result in extinction of the free-ranging population in the next 6 years, if it is not supplemented by further releases from the captive population.

Major difficulties faced

- Elephant damage to the fences of the enclosures was a constant problem.
- In the enclosures, juvenile hartebeest were killed by leopards, which had to be caught and relocated.
- Lions were a serious threat because if they managed to access the enclosures they invariably killed adult hartebeest, which had a greater impact on growth of the population than the loss of juveniles. In the process of trying to remove a lioness from the first enclosure, Malilangwe's Wildlife Manager was mauled.
- Exposure during cold, wet periods in the dry season, when the hartebeests' reserves were at their lowest, was an infrequent but significant cause of mortality.
- Adult bulls are very aggressive, and will fight and kill subadult bulls (>I year) if these are not removed from the enclosure.
- Balancing the requirements of a photographic tourist operation that relies heavily on frequent sightings of the large carnivores, and the needs of a small re-introduced antelope population that is highly sensitive to predation, is a particularly difficult management problem. Capon (2011) showed that a lion

density in excess of 0.05/km^2 appeared to be unsustainable for a population of sable antelope at Malilangwe. This may also be true for the hartebeest population. The current lion density at Malilangwe is 0.07/km^2, but it has been as high as 0.1/km^2 in the past.

Major lessons learned

- Successful re-introduction of a low density antelope is only possible under conditions of low predation. To establish a self-sustaining, free-ranging population of hartebeest at Malilangwe the lion density should possibly be <0.05 km^2.
- To achieve rapid growth of the breeding nucleus there should be sufficient breeding animals (>30) to ensure that the population is positioned within the exponential phase of the logistic growth curve. With an initial nucleus of only 6 animals, it took the population in the second enclosure five years to enter an exponential growth phase.
- A massive outbreak of anthrax occurred at Malilangwe in 2004. Despite mortality of several species in the enclosures, no hartebeest succumbed to the disease. Hartebeest appear to be particularly resistant to anthrax.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- The re-introduction can only be considered partially successful because the established free-ranging population is currently not self-sustaining, with extirpation being prevented only by periodic supplementation from the captive population.

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An investigation into the effect of individual personality on re-introduction success, examples from three North American fox species: swift fox, California Channel Island fox and San Joaquin kit fox

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Introduction

Swift fox (*Vulpes velox*) and San Joaquin kit fox (*Vulpes macrotis mutica*) are small, den-dwelling foxes found in North American arid grassland habitats. The Catalina Channel Island fox *(Urocyon littoralis catalinae*) is endemic to Catalina island. Swift fox (SWF) are listed by the IUCN as Least Concern, but listed in Canada as Threatened (COSEWIC, 2009) and as a Category One species under the US Endangered Species Act (USFWS, 2001). The kit fox is listed as Least Concern, however the sub-species San Joaquin kit fox (SJKF) is listed as Endangered under the US Endangered Species Act (USFWS, 1998). The Catalina Channel Island fox (CCIF) is listed as Critically Endangered (IUCN, 2008). Decline of the species' were associated to anthropogenic factors. Extreme habitat loss caused a loss of >90% the historic range of both SWF and SJKF, with decline also attributed to non-target predator control, whilst CCIF numbers crashed due to an outbreak of distemper. As a result, there have been on-going recovery efforts for each species, incorporating captive breeding and re-introduction (SWF and CCIF), habitat restoration (SWF, SJKF and CCIF) and re-introduction feasibility planning (SJKF).

San Joaquin kit fox

Goals

- Goal 1: Development of methods to determine individual personality type via an assessment of boldness.
- Goal 2: Demonstrate repeatability of tests to show individual stability of personality type.
- Goal 3: Monitor assessed individuals either post-release or for 1+ years post-behavioral assessment.
- Goal 4: Evaluate the effect of personality on survival post-release and the effect of habitat differences on population-level personality and fitness variables.
- Goal 5: Identification of optimal levels of boldness relative to habitat conditions.

Success Indicators

- Indicator 1: Calculation of a boldness score for each assessed individual.
- Indicator 2: Evaluation of within population personality variation.
- Indicator 3: Evaluation of between population personality variation.
- Indicator 4: Obtaining post-release/post-assessment survival and fitness data on a sample size that allows statistical determination of a relationship between personality and survival, movement and reproductive output.

Project Summary

Feasibility: One factor affecting the success of re-introduction is intraspecific behavioral variation. The existence of different personality types, e.g. boldness, indicates adaptive strategies within a species that are acted on by natural selection (Wilson & Richards, 2000). Inappropriate boldness levels may have deleterious effects on fitness. With levels of boldness subject to natural selection, it is possible that release candidates with optimal levels of boldness for a source-habitat similar to the release site may be more likely to survive than individuals from a source population with differing selection pressures than the release site. However, it is also likely that behavioral variation is key within a release population to ensure the ability of the founders to adapt to environmental pressures. Developing an assessment of the likely behavioral response of an individual to re-introduction was the overall goal across the three projects. The aim was to find a means of predicting the likely response of an individual to the novelty of release via a simple behavioral personality test, and implement this knowledge to improve survival and re-introduction success. The first project assessed personality of individual captive-bred SWF and survival after release at a site with predation (Bremner-Harrison *et al.*, 2004). The second project assessed the effect of personality on survival and reproductive output of CCIF released in an environment with no predators (Bremner-Harrison *et al.*, 2005). The final project evaluated the effect of personality in two free-living populations of SJKF in the San Joaquin valley of California, USA, with differing habitats (urban and rural). These two populations are possible source populations for a planned re-introduction.


Fox trapping in Lokern

Implementation: A boldness score was calculated for 34 captive-bred SWF using two repeats of a four x novel-object test prior to release. A boldness score was calculated for 11 captive CCIF using a reduced test of a two x novel-object test. Three measures of boldness were calculated for free-living SJKF: a handling boldness score assessed during trapping and handling (T/H); an extended novel object test (ENOT: two x novel object (1 novel food and 1 novel threat) + two x baseline assessing pups) and a rapid novel object test (RNOT: 1 x novel-object test assessing juveniles and adults).

Post-release monitoring: Thirty-one SWF were released onto the Blackfeet Indian Tribal Reservation in Montana, USA; 16 were radio-collared. Foxes were monitored intensively for 6 weeks, weekly/fox for the following 4.5 months, and again intensively for a 2-week period 6 months post-release to determine survival. Nine radio-collared CCIF were released in the final year of a four-year re-introduction program on Catalina Island. Post-release monitoring was conducted twice-weekly/fox. For SJKF the T/H test assessed 87 urban:67 rural. The ENOT assessed 24 urban:9 rural, with 21:1 radio-collared. The RNOT assessed 27 urban:27 rural, all radio-collared. Post release monitoring for ≥1 year post-boldness testing aimed to locate each SJKF a minimum of once per week, recording survival, dispersal data, and reproductive data. The relationship between the variables listed and boldness was assessed for each fox species.

In a habitat with predators it was determined that SWF that died (n = 5) were those with higher levels of boldness (t_{14} = 2.942, P < 0.01). Boldness scores were positively correlated with the total distance moved from release site (r_{14} = 0.588, p<0.02) and the mean distance moved per telemetry fix (r_{14} = 5.574, p<0.02), the mean distance moved between fixes was significantly greater for foxes that died (U = 6, p<0.02).

In a predator-free habitat released CCIF had no mortalities. However, CCIF showed a trend towards foxes with higher boldness having increased reproductive output. For SJKF, the urban habitat had no predation but vehicle-caused mortality and food resources were consistent, but there were limited dispersal opportunities. The rural habitat had predation risks, food resources fluctuated and dispersal opportunities were unlimited. Overall, urban foxes were bolder than rural foxes (T/H: t_{149}=2.52, P<0.01; ENOT: t_{29}=3.05, P<0.005; RNOT:

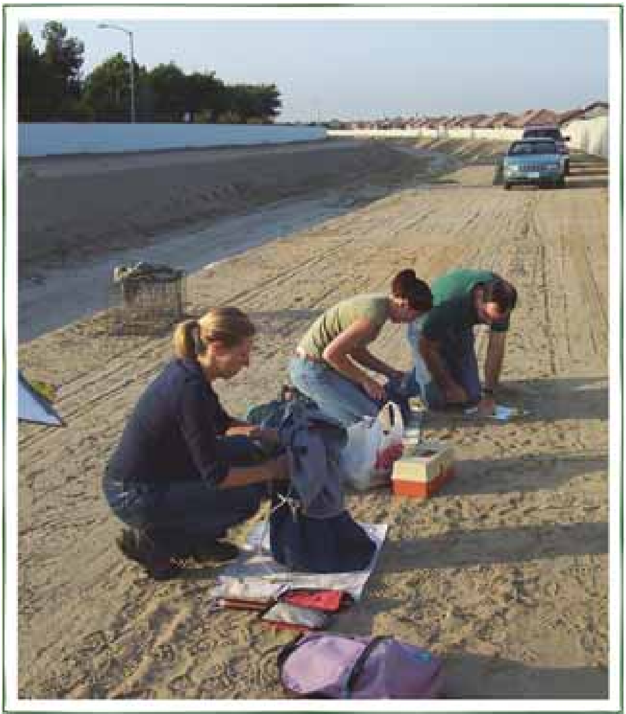
t_{43}=2.85, P<0.01), and there were a wider range of boldness scores across urban foxes, indicating a greater variability in the expression of behavioural type. Urban adults and juveniles were bolder than rural (T/H: t_{63}=2.15, P<0.05; RNOT: t_{17}=2.30, P<0.05), but there was no difference between urban vs. rural pups (T/H and ENOT test for baseline and novel threat). Within populations there was no difference between boldness across ages classes of urban foxes but rural adults were less bold than rural juveniles and pups (t_6=-2.12, P=0.07). The differences in boldness between and within habitats across age classes indicate young bolder foxes are more likely to be selected against than their shyer conspecifics. Rural foxes that died had higher boldness than surviving foxes (t_{33}=-2.01, P=0.05), with a similar trend in urban foxes. However, bolder rural foxes that survived to reproductive age reproduced in their first year (t_{17}=2.058, P=0.05) and had increased litter sizes (F_{18}=4.729, P<0.05).

Major difficulties faced

- Limited data: In the SWF project 31 foxes were released, 16 of which were radio-collared. Survival and movement was not obtained for the remaining 15 foxes as the stakeholders in the project did not wish trapping to be conducted at the site during the period of the study.
- Appropriate tests for free-living animals: The behavioral tests were originally designed for use on captive animals. Conducting the tests on free-living individuals highlighted the need for modification of the tests to produce robust data on wild animals.
- More intensive monitoring required: While the three studies produced informative insights into how personality can potentially influence re-introduction success, more intensive monitoring of individuals would produce greater information about dispersal movements and resource use. Limited numbers of personnel in the SJKF project meant that there was a limit to the amount of time that could be spent monitoring each fox. In addition, there were a large number of behavioral tests to be conducted in the short period of time before pups dispersed from the natal den, however in the interest of consistent evaluation of behavioral type only one observer conducted the test which limited the number of dens that could be observed.
- Limited animals for testing: As the behavioral analysis was conducted as part of ongoing conservation efforts, the number of animals available for testing were dependent on either the numbers captive-bred or the numbers available to be tested, i.e. number of individuals born or trapped. While the numbers available produced informative data regarding the effects of personality on re-introduction, in some instances (especially CCIF) data collected was not of the magnitude to conduct statistical analysis. While this does not restrict disseminating information to conservation organizations via project reports, it limits the capacity to disseminate to a wider audience through scientific journals.

Major lessons learned

- Boldness levels of individuals may be inappropriate or put the individual at a disadvantage for habitat conditions: Foxes that had very high levels of boldness showed higher levels of mortality due to both predation and vehicle-



Processing foxes trapped along water canal in Bakersfield

associated mortality than those with lower boldness. Predicting the likelihood of individuals to survive relative to the mortality risk at a release site may allow for determining either the placement of individuals within the site or levels of pre-release training required. While it is recommended that a range of behavioral types be released, it may be advisable in particularly high risk situations to delay releases of particular individuals until the predator density is reduced if deemed appropriate.

- Levels of boldness may influence the movement of an individual within a release site: Very bold foxes showed a greater propensity for either increased total movement (SJKF) or for moving around more between daily radio-telemetry fixes (SWF). This resulted in bolder foxes either leaving the release/study site or having increased mortality associated to increased movement. This was likely due to foxes that moved around more having a greater risk of encountering a predator, or alternatively being in unfamiliar terrain with limited knowledge of escape dens. Therefore it was concluded that foxes with overly high levels of boldness may be at a disadvantage when released at a site with high predation risk. The movement data suggests it is likely that individuals of particular personality types will explore their new release site in different ways. For example, bolder individuals may explore the release site more thoroughly and be more likely to disperse out of the site, but may also be more likely to find food resources and potential mates. Shyer individuals may remain closer to their release site, expose themselves less to mortality risks but may be less likely to locate adequate resources and potential mates. Intensive monitoring would have provided information regarding these theories; however current and future studies are focused on exploring these areas to determine whether it is possible to produce a release placement strategy in regards to resource availability and predator risk based on personality type.
- Bolder animals that survived were those that reproduced in their first year and had bigger litters: This suggests there is a trade-off between high boldness having a greater risk of mortality but the potential for greater reproductive output. Therefore, while bolder animals may be more likely to be predated on

or put themselves at risk relative to anthropogenic threats these individuals are also more likely to facilitate recruitment in the founding release population faster than shyer individuals.

- Captive-bred SWF litters and SJKF litters in both the urban and rural habitat showed variation in boldness levels: This supports the view that boldness is an adaptive trait with variation in the litter allowing a greater potential for there being individual personality types in the litter who will be suited to current environmental conditions. The reduction in boldness levels in the higher age classes indicates bolder SJKF were selected against at a young age. However, the continuing presence of variation in litters suggests it is an adaptive strategy to continue to produce variation in pups. Therefore, it is recommended that a founder population contains representatives of all behavioral types to ensure the potential for the population to adapt under variable environmental conditions, or if there is variation within the release site. It may be advantageous in some instances to consider releasing founders with behavioral types in certain proportions depending on habitat pressures, and then varying the mix in subsequent releases to ensure variability. For examples, in release sites where predators are present the initial releases could comprise of a majority of shy individuals, but gradually switch to a higher proportion of bolder individuals as density increases and the need for dispersers to create linkage with neighboring populations arises.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- Co-operative partnerships and financial support: A broad range of organizations supported these projects. The SWF project was facilitated by the Cochrane Ecological Institute, Defenders of Wildlife, the Blackfeet Fish, Game and Wildlife Department in Montana and Queen's University of Belfast. Financial support was provided by The Department of Education for Northern Ireland. The CCIF project was a co-operative effort between the California State University - Stanislaus Endangered Species Recovery Program and the Catalina Island Conservancy. The former provided financial support and personnel while the latter provided access to animals and logistical support such as accommodation and transport. The SJKF project was conducted by personnel from the California State University - Stanislaus Endangered Species Recovery Program with funding provided by the Central Valley Program Conservation Project (CVPCP), a funding source administered by the US Bureau of Reclamation.
- Local community support: Both the SWF and the CCIF project experienced high levels of support from the community. Community feeling regarding SJKF was considered varied within the urban habitat; however, the majority of people encountered were supportive. The SWF project in particular demonstrated community support through the cooperation of the Blackfeet

Indian tribal community. Swift fox are considered a culturally significant animal to the Blackfeet, therefore they were highly supportive in terms of allowing access to land, passing on information regarding fox sightings, den locations and assisting with monitoring.

- Development of behavioral ethograms and boldness tests: Initial work was conducted on captive individuals. While working on captive foxes was a function of the re-introduction process for SWF, it allowed for effective development and modification of the behavioral tests. Close range observation of foxes habituated to the observer allowed for development of an extensive ethogram that was then modified for the CCIF and SJKF projects.
- Adaptation of tests for use in the wild: The original behavioral test was developed for captive foxes and relied upon the individuals remaining in a fixed location for the 1-hour testing periods. This method was generally applied successfully to SJKF pups who remained within the immediate vicinity of the natal den during the testing period. However, difficulties were encountered with locating natal dens between tests as adults move pups between den sites as a means of predator avoidance and as fleas build up in dens. Intensive monitoring of parents allowed all pups to be located and behaviorally tested. The original test was not particularly suited for juvenile and adult SJKF as shortly after emerging from the den individuals would leave to hunt. The CVPCP provided an extension of contract and funding to allow for effective testing of adults and juveniles using a modified behavioral testing method.

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Establishing new populations of European mink in Hiiumaa and Saaremaa Island, Estonia

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Introduction

European mink (*Mustela lutreola*, Linnaeus, 1761) is a small semi-aquatic mustelid inhabiting the banks of forested rivers and streams. Its historical range covered most of the European continent except Scandinavia and parts of Balkans. Over the last 150 years its original range has reduced drastically. The current range consists of few isolated and shrinking fragments (Maran, 2007 and references therein). There is no data available about any extant viable wild population. The main factors operating the extinction are i) habitat loss, ii) over-exploitation and iii) impact of alien American mink (Maran *et al.*, 2011). In Estonian legislation the species belongs to the first and the most strict protection category. Since 2011 it is listed in IUCN Red List as Critically Endangered species. It is legally protected in all range states. In European Union it is listed in appendixes II and IV of the Habitat Directive and considered to be a priority species of the Community. The release operation has been conducted in two islands: Hiiumaa (1,000 km^2) and Saaremaa (2,400 km^2).

Goals

- Goal 1: Remove the American mink (*Macrovison vison*) feral population in Hiiumaa Island.
- Goal 2: Establish European mink island population in Hiiumaa Island up to the size of assessed post-winter carrying capacity: 50 - 92 mink.
- Goal 3: Improve the riparian habitats in Hiiumaa Island to increase the post-winter habitat carrying capacity to 88 – 109 mink.
- Goal 3: Establish European mink island population in

European mink


Typical mink habitat

Saaremaa Island up to the size of assessment carrying capacity of 150 – 300 mink.

Success Indicators

- Indicator 1: American mink removed from Hiiumaa Island.
- Indicator 2: Breeding wild population: size close to estimated carrying capacity in Hiiumaa (Dagö) Island.
- Indicator 3: Breeding wild population: size close to the estimated carrying capacity in Saaremaa (Ösel) Island.

Project Summary

Feasibility: The estimation of the carrying capacity based on the pilot studies were conducted in both island (unpublished). For Saaremaa Island a second pilot study was conducted to assess the capacity of core areas instead of the entire island. The post-winter carrying capacity of three core areas was assessed to be around 58 – 73 mink. The base for these release operations has been the conservation breeding of the European mink in Tallinn Zoological Gardens (in the frame of the EAZA EEP program), and breeding facility there hosts around 100 mink.

Implementation: The first releases on both islands were regarded as experiments to evaluate the feasibility of the operation. The release in Hiiumaa Island has been conducted yearly since 2000. In total, 475 (2012) mink has been released there:

- In 2000 - 2001, hard release combined with preconditioning was tried. The mortality of released animals was excessively high.
- In 2002 - 2003, the feasibility of pregnant female's release was tested as a mean to achieve a fast increase in wild-born mink. The females survived, but the litter disappeared at age of around two months.
- Since 2004 only yearlings born in release enclosures in Hiiumaa Island and in Tallinn Zoo have been released.
- In 2000 - 2003 the released mink were radio-collared to collect information about their post-release behavior.
- In 2012, a pilot release was conducted in Saaremaa Island; 11 radio-collared mink were released there and their post-release behavior was followed for two months.

Several studies have resulted from Hiiumaa operation:

- Maran *et al.* (2009) found that the mortality of released animals was the greatest during the first 1 - 1.5 month, the males mortality was lower than that of the females and that the main factor causing mortality were larger predators, like fox, stray dogs and bird of prey.
- Põdra *et al.* (2012) found that the atypical food prevailing immediately after the release in mink diet was substituted to typical wild mink diet within 30 days.
- The post-released movements of mink were analyzed in Harrington *et al.* (submitted 2012).

The release operation in Hiiumaa was regularly highlighted in local and national mass-media. The public awareness study conducted in 2004 (unpublished) revealed very high awareness among locals (97%) and highly positive attitude (>85%) to the project. Number of spawning-ponds for common amphibians have been excavated close to mink habitats to mitigate the negative effect land reclamation activities may have to important prey species. In addition, in collaboration with government agencies, the habitat quality of stream habitat was improved in several locations.

The main concerns have been the following:

- Suitable habitats patches in the island are scattered and none of them forms a sufficiently large source habitat for mink. As a solution, a running-water habitat improvement project was planned by the State, but is likely to be abandoned due mismatch between various formal governmental procedures.
- The low level of breeding in the wild.
- EU demanded anti-rabies vaccination campaign, which obviously hyper-increased the abundance of medium-sized carnivores and is suspected to negatively effect the establishment of the European mink population.
- The pilot release in Saaremaa Island (2012) was not promising as seven mink of 11 mink died within the first month after release and 71% of these were killed by fox or other predators. The following conclusions have been drawn: i) selected release site was suboptimal, ii) the attitude of local inhabitants to the project was highly positive, and iii) the further release operations may not be feasible due to the high abundance of foxes.


Captive bred European mink

Post-release monitoring: In Hiiumaa Island yearly monitoring started in 2000

Field researchers working on releasing mink

after release. There has been more or less steady increase in size of the post-winter population size (around 30 - 35 animals) and breeding in the wild has been observed. However, the demographic structure of the population is biased towards old and/or released animals. The causes of insufficient reproduction in the wild remain unclear. It is suspected that either the spatial structure with no compact source habitat area available or the male's abnormal mating behavior (Kiik *et al.*, 2013) might be behind it. The comparison of various monitoring techniques (track counting, live-trapping, mink-rafts and trail cameras) has raised the issue of reliability of data collected under different monitoring schemes.

Major difficulties faced

- Unstable funding complicates the planning and performance of the operation.
- Unforeseen negative factors like the rabies vaccination campaign jeopardizes the operation.
- The spatial structure of the habitat distribution in the island is important factor and this was not taken into account in the feasibility study.
- Inflexibility of state conservation procedures is incompatible with species conservation action needs.

Major lessons learned

- The flexibility and open decision-making over the actions of the operation is crucial.
- Positive attitude of local inhabitants largely depends upon information flow, the personal contacts with local opinion leaders are critical; with time it is more complicated to keep the public information flow consistent as different stakeholders of the project will share information from their perspectives and needs.
- The administrative and political considerations of wider scope, but with serious implications to the outcome release operation are difficult of mitigate.
- Instead of overall carrying capacity of the island the core areas have to be evaluated. Scattered habitat patches without one strong source habitat are likely to result in lower than expected level of reproduction.

- Conventional single-species approach must be replaced to more holistic approach with attention to relations to other species, to the habitat and human interactions.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- Good team-work and very tight connections between *in situ* and *ex situ* teams.
- Rigid government procedures cause serious delays in the operations.
- Unstable funding will result in less effective operation.

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Re-introduction of greater one-horned rhino in Manas National Park, Assam, India - *under the Indian Rhino Vision 2020*

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Introduction

The greater one-horned rhino (*Rhinoceros unicornis*) is listed in IUCN Red List as Vulnerable while in CITES as Appendix I. The species is also included in Schedule I of the Wildlife (protection) Act 1972 enacted by the Government of India ensuring high priority in conservation of the species. The greater one-horned rhino once ranged throughout the entire stretch of the Indo-Gangetic Plain of the Indian subcontinent, along the Indus, Ganges and Brahmaputra river basins, from Pakistan to the Indian-Burmese border, including parts of Nepal, Bangladesh and Bhutan but excessive hunting reduced their natural habitat drastically. Today, about 3,250 rhinos live in the wild in India and Nepal out of which 2,500 are found in India's Assam alone. They prefer the alluvial plain grasslands of the Terai and Brahmaputra floodplain. As a result of habitat destruction and climatic changes their range has gradually been reduced and today, their range has further shrunk to a few pockets in southern Nepal (Chitwan National Park (NP), Bardia National Park and Sukhlaphanta Wildlife Reserve), Uttar Pradesh of India (Dudhwa NP), northern Bengal (Garumara National

Greater one-horned rhino

Rhino captured and collared at Pabitora for release in Manas National Park

Park and Jaldapara Wildlife Sanctuary) and the Brahmaputra Valley (Kaziranga National Park, Orang National Park, Manas National Park and Pabitora Wildlife Sanctuary).

Goals

- Goal 1: Range expansion of the rhino to potential habitats within Assam by the year 2020 through wild to wild translocation.
- Goal 2: To increase the rhino population in the wild to about 3,000 in Assam by 2020.
- Goal 3: To conserve existing grassland habitats in Assam through re-introduction of rhinos.

Success Indicators

- Indicator 1: How many rhinos captured at donor rhino bearing areas and translocated to recipient sites within Assam.
- Indicator 2: Rhino population increase in rhino bearing areas in Assam.
- Indicator 3: Infra-structure enhancement in recipient rhino bearing sites.

Project Summary

Feasibility: The Indian Rhino Vision 2020 was initiated in 2005 which is a collaborative initiative of Assam Forest Department, Bodoland Territorial Council, International Rhino Foundation, World Wide Fund for Nature and U.S. Fish and Wildlife Service. A rhino task force was constituted in 2005 to promote the plan of Indian Rhino Vision 2020. Since 2005 until April 2008 feasibilities of enhancing habitat and security to cater the need of rhinos in the recipient sites were thoroughly assessed and followed up with infra-structure development to ensure that the recipient site is capable monitor them to protect translocated rhinos and habitats. Interaction with fringe communities of the recipient sites were undertaken to generate and enhance their awareness on conservation issues, specifically on rhinos to build up community support to ensure the future of the rhinos. An indepth security assessment was carried out in the recipient sites along with habitat assessment to ensure that ground is set to receive translocated rhinos.

Implementation: After habitat and security assessment was done and report submitted to the Rhino Task Force of Assam, for rhino translocation within Assam which is one of the key output of the IRV 2020, a Translocation Core Committee

Monitoring released rhino on motorbikes

(TCC) was formed by the Rhino Task Force to initiate steps to start rhino capture and Translocation. After improving the security infrastructure in Manas, the first batch of rhino translocation was planned in April 2008 from Pabitora WLS to Manas NP. Accordingly two male rhinos were captured in Pabitora WLS and translocated to Manas NP. Both the rhinos were radio collared. The initial plan is to capture and translocate 20 rhinos to Manas NP of which 10 rhinos each to be captured from Pabitora WLS and Kaziranga NP and translocated to Manas NP. Since April 2008 until March 2012, 18 rhinos were captured and translocated to Manas NP of which 10 were captured from Pabitora and 8 from Kaziranga. Rhinos were captured during morning and captured rhinos were put into a wooden crate and then loaded into Truck for transportation to Manas NP whih is the recipient site. The trucks carrying captured rhinos move from the capture sites (Pabitora WLS and Kaziranga NP) to release site (Manas NP) in the evening so as to reach Manas NP by early morning. Rhinos are then released in the wild in Manas NP in early morning.

Post-release monitoring: All the rhinos released in the Manas NP under IRV 2020 are fitted with VHF radio collars and are being monitored daily by a dedicated team using telemetry equipment. The monitoring is carried out round the clock using vehicles, trained elephants, motorcycles, etc. and after initial tracking of the rhinos, attempt is made to physically observe them. All the rhinos are also ear notched and physical identification is done with the help of their unique identification marks. The monitoring efforts are recorded using pre-designed formats and is entered into a GIS platform for analysis and outputs and annual monitoring reports are produced.

Major difficulties faced

- Procurement of tranquilizing drugs from abroad.
- Unpredictable weather.
- Keeping the team motivated specially the team involved in patrolling and monitoring in Manas.
- No dedicated manpower for the program, skilled and experienced persons of the state offer voluntary service as such timelines needs to be flexible.

Major lessons learned

- Good understanding and team work.
- Plan in advance and execute the plan within timeline.
- Procurement of tranquilizing drugs often takes more time than expected due to complicated import procedures in India.
- Short expiry time of the imported tranquilizing drugs and as such time bound capture using the drugs is important before drugs get expired.

Success of project

Highly Successful	Successful	Partially Successful	Failure
√			

Reason(s) for success/failure:

- Good coordination and team spirit to make it success.
- Commitment of the Government and other partners to make it success.

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Re-introduction of tule elk to Point Reyes National Seashore, California, USA

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Introduction

Tule elk (*Cervus elaphus nannodes*), a subspecies endemic to California, was historically found in large herds throughout much of central and coastal California. Market hunting during the California Gold Rush decimated these herds, and by 1895, only two to 10 elk remained. This remnant group was protected and served as the source for early relocation efforts (McCullough, 1971). Early efforts were generally unsuccessful but did establish a herd in California's Owens Valley, outside their historical range, in 1933. The herd grew rapidly and supported six controversial hunts between 1943 and 1969. In an effort to limit hunting, concerned preservationists formed the Committee for the Preservation of Tule Elk in 1960. Public pressure resulted in the California State Legislature passing a law in 1971 that halted hunting until either state-wide numbers reached 2,000, or no further unoccupied elk habitat existed. This law prompted the California Department of Fish and Game to begin re-introducing tule elk throughout their former range. In 1976, the U.S. Congress passed a resolution that concurred with state law and directed federal agencies to make lands available for re-introductions within the subspecies' historical range. Point Reyes National Seashore was identified as a potential translocation site.

Male elk with Pacific Ocean in the background © Jeff Wilson

Goals

Tomale Point from the air © Reg Barrett

- Goal 1: Establish and maintain viable populations of tule elk within the subspecies' native range at Point Reyes National Seashore, California.
- Goal 2: Manage tule elk using minimal intrusion to regulate population size, where possible, as part of natural ecosystem processes.
- Goal 3: Provide for a free-ranging tule elk herd at Point Reyes National Seashore.
- Goal 4: Research and monitor the elk populations and their habitat over time.
- Goal 5: Provide the public with interpretation and information on tule elk conservation biology and management.

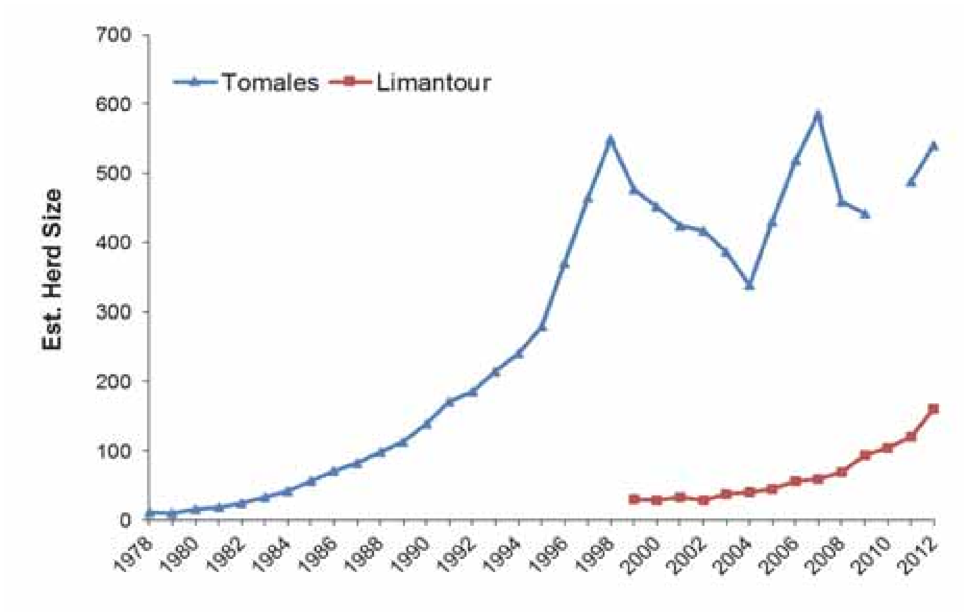
Success Indicators

- Indicator 1: Re-introduced tule elk do not experience extirpation or become exposed to the threat of extirpation.
- Indicator 2: Re-introduced tule elk are allowed to become self-regulating to the extent possible.
- Indicator 3: A tule elk herd not restricted in their movements by fencing is successfully established outside the original fenced release site.
- Indicator 4: Research and long-term monitoring of tule elk and their habitat at Point Reyes are incorporated in to an adaptive management program.
- Indicator 5: Tule elk are readily accessible to the public for viewing and information on their life cycle and conservation status is available to the public.

Project Summary

Feasibility: Point Reyes National Seashore (288 km^2) historically supported more than 1,000 tule elk, as indicated by biological specimens and historical accounts (McCullough, 1971). By the mid-1800s, tule elk were extirpated from Point Reyes due to agriculture, logging and market-hunting. In 1976, Point Reyes was identified as a potential re-introduction site in state and federal conservation plans. It was decided that the ideal tule elk release site within Point Reyes needed to be a confined area separated from domestic cattle operations where the re-introduced elk could be held in a small acclimation pen prior to release (i.e., soft release). The northernmost 10.3 km^2 of the Point Reyes Peninsula (Tomales Point) met these criteria: a 5 km fence at the base of the Point could confine elk and the existing cattle grazing lease for the area had expired.

Fig. 1. Showing exponential growth of the population



Implementation: Ten tule elk (2 males:8 females) were translocated to a holding pen at Tomales Point in March, 1978. These 10 elk and seven additional calves born in the pen (4 males:3 females) were released in September 1978, and shared the range with cattle until the following year. The re-introduced elk seemingly flourished during the first few months following release. However, by mid-summer 1979, one adult male died and a second adult male was severely emaciated and was removed from the range. Both males had deformed antlers. The females and calves were provided supplemental feed from September 1979 to April 1980 to alleviate apparent malnutrition evident from emaciation and light-colored brittle pelage. Two female elk born in the pen in 1978 were culled in March 1980 after exhibiting emaciation and severe diarrhea. In an effort to supplement the genetic diversity of the original re-introduced herd, an additional three adult males were re-introduced to Tomales Point in December 1981, but these elk disappeared in early 1982.

Post-release monitoring: Tule elk at Point Reyes have been monitored closely since their re-introduction. The severe diarrhea observed in the herd shortly after their re-introduction was determined to be the result of Johne's disease (*Mycobacterium avium paratuberculosis*) (Jessup *et al.*, 1981), and the light colored pelage and antler deformities were attributed to trace element deficiency (Gogan *et al.*, 1989). Following an initial period of slow growth, the herd exhibited approximately 20 years of exponential growth (Cobb, 2010) (Figure 1). By 1998, the herd numbered approximately 450 animals (4 elk/ha).

The rapid herd growth at Tomales Point raised concerns over potential negative impacts that a high density elk herd may have on native flora and fauna. Additional concerns were raised over the potential for the expanding elk population to overshoot the area's ecological carrying capacity, leading to habitat degradation, and then crash to a lower abundance, resulting in a secondary genetic bottleneck. In response to these concerns, the National Park Service considered alternative measures that included no action and various combinations of actions including culling, contraception, translocations, and removal of the elk fence at Tomales Point (National Park Service, 1998). Ultimately, a decision was made to evaluate the effectiveness of translocation and artificial population control methods within the framework of adaptive management. Between 1997 and 2001, 30 to 50 elk cows at Tomales Point were contracepted annually with porcine zona pellucida (PZP), which effectively prevented pregnancy for one year. In 1998, biologists captured and translocated 45 elk cows and bulls from Tomales Point to an unrestricted region of Point Reyes (Limantour) following a test and cull screening program for Johne's disease, thereby establishing a free-ranging herd beyond Tomales Point.

Following this management intervention, elk at Tomales Point exhibited periodic swings in numbers from 1998 to 2012, suggesting that the herd may have reached a stochastic carrying capacity (Figure 1). Shortly after release at Limantour, two to three elk moved approximately 10 km from the release site and established a second free-ranging herd (D Ranch) on cattle ranchlands. The newly established herds initially exhibited a pattern of early population growth similar to that observed at Tomales Point, but then began to increase rapidly. With abundant forage resources, these newly established herds are predicted to increase exponentially (Cobb, 2010). Rapid growth of the D-Ranch herd combined with the elk's habitat preferences has caused concern among cattle ranchers within the Seashore over elk use of forage resources, which may escalate as elk numbers increase (Cobb, 2010).

Major difficulties faced

- Trace element deficiency and Johne's disease threatened the initial success of the Point Reyes tule elk re-introduction.
- Male tule elk introduced to Tomales Point to enhance the genetic diversity of the initial herd did not survive.
- Presence of Johne's disease made tule elk at Point Reyes

Ranger and female elk at Tomales Point
@ McCrea Cobb


Visitors viewing elk at Tomales Point © NPS

unsuitable as a source for future state-wide re-introductions, as dictated by state law, and required screening (test and cull) prior to establishing a new free-ranging herd within Point Reyes.

- The potential threat of disease transmission to the insular tule elk at Point Reyes, specifically Chronic Wasting Disease (CWD), made it difficult to supplement the herd to mitigate the potential for inbreeding depression.
- A lack of effective population-limiting predators at Point Reyes allowed for rapid irruptive growth of the elk herds. The limited area available to elk at Tomales Point led to a short lived fertility control program and one time removal of elk for relocation. The expansion of the free-ranging Limantour herd has led to concerns among local cattle ranchers.

Major lessons learned

- Screen for nutritional conditions within domestic and wild species at release site prior to the re-introduction. Trace element deficiency in elk at Tomales Point was likely unavoidable due to naturally low levels of certain trace elements (copper and selenium) in the underlying soils and bedrock. Screening for diseases may not be practical as the presence of many diseases cannot be determined by screening.
- Identify potential population regulation factors prior to re-introduction. Knowledge of potential population limiting factors (or the lack thereof) may allow predicting population growth based upon outcomes elsewhere and thereby allow identification of potential future management actions.
- Address means of alleviating potential inbreeding depression.
- Recognize the likelihood of public involvement in future management actions and identify a socially acceptable means of any possible future population control prior to re-introduction.
- Quantify resource selection to predict areas of likely range expansion and potential conflicts with human activities prior to a re-introduction. This may allow for proactive human-wildlife conflict management.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- A self-sustaining herd of tule elk was established at the original release site at Point Reyes and a second free-ranging herd was established using progeny from the first re-introduction.
- Concerns over the potential transmission of diseases to and from tule elk at Point Reyes blocked plans to manage the herd as part of a meta-population.
- For the first 20 to 30 years post re-introduction, potential conflicts between tule elk and cattle ranching operations have been minimal, despite their close proximity.
- Tule elk at Point Reyes have displayed no outward signs of inbreeding depression, even though they have some of the lowest genetic diversity of any tule elk herd
- Ready opportunities for viewing of tule elk at Point Reyes enhance public understanding of the area's rich natural history.

Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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Re-introduction of Arabian gazelles in a fenced Protected Area in central Saudi Arabia

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Introduction

Historically, the Arabian gazelle (*Gazella arabica*) locally known as *Idmi* occurred across most of the Arabian Peninsula from the Araba Valley in southern Israel, along the Hejaz and Asir Mountains in western Saudi Arabia, through Yemen, Oman and into the Emirates. In Saudi Arabia the Arabian gazelle population has been declined dramatically throughout its range since the middle of the 20th century (Habibi 1986), and the IUCN Red List (IUCN 2012) currently ranks this species as 'Vulnerable' (A2ad).

Small relict populations of Arabian gazelles used to occur in Al Khunfah and Harrat al Harrah in the north of Saudi Arabia (Wacher, 1993; Seddon *et al*., 1997), and on the Tihama coastal plain in Wadi Hali (Islam pers. obs.) 80 km south of Al Qunfidah, while animals recorded near Al Farah (Boug *et al*., 2012). On the Farasan Islands a strong population of about 1,000 individuals survived (Wronsky *et al*., 2011, 2012), and in two protected areas (Ibex Reserve, Uruq Bani Ma'arid) Arabian gazelles were released from 1990 to 2007 (Islam *et al*., 2012). Most records of natural Arabian gazelle populations in Saudi Arabia originate from the western part of the Kingdom, i.e. the Asir, Sarawat and Hejaz Mountains.

Historically Arabian gazelle used to occur in Mahazat as-Sayd Protected Area in central part of Saudi Arabia and were exterminated by anthropogenic and other pressures. Since the Arabian gazelles presence was confirmed from interviews of local people. The Strategy and Action Plan of the National Wildlife Research Center (NWRC) suggested the re-introduction of Arabian gazelle (Islam *et al*., 2009) should occur and 40 (12 males:28 females) captive -bred animals were successfully released in 2011 & 2012. This project is particularly significant as it is one of the first successful releases for the

Arabian gazelle at night © M.Z. Islam

species in over twenty years. After many years of dedicated work to identify and conserve different species of gazelles in Saudi Arabia, it was successfully released. The release is part of the ongoing efforts in the Kingdom to conserve a variety of antelopes, an initiative that is strongly supported by the Saudi people.

Arabian gazelles release in MZT by H. H. Prince Bander © D. Kifle

Goals

- Goal 1: To re-establish wild and self-sustaining populations of Arabian gazelle in Mahazat as-Sayd Protected Area in Saudi Arabia.
- Goal 2: Manage the re-introduction of the herds in the protected areas.
- Goal 3: Re-introduce the animals in suitable habitats.
- Goal 4: Study the ecology and biology of the Arabian gazelle in protected area.
- Goal 5: Balance between grazing and browsing animals in Mahazat.

Success indicators

- Indicator 1: Healthy breeding Arabian gazelle population in Mahazat as-Sayd Protected Area.
- Indicator 2: The captive herd at KKWRC is maintained for re-introduction programs for other protected areas.
- Indicator 3: The re-introduction of Arabian gazelle in Mahazat for more than two years, which now has a breeding population and considered to be a partial success.
- Indicator 4: Productivity by wild Arabian gazelles high.
- Indicator 5: Society and government supports re-introduction and Mahazat has been suggested for national and international tourists.

Project Summary

Feasibility: Arabian Gazelles were previously occurred in Mahazat area (22° 15'N - 41°40'E), which is tract of open desert steppe habitat of tropical and arid climate with gentle topography in southwest of Saudi Arabia c.150 km northeast of Taif. Historically the species had been extirpated, primarily by excessive hunting. After the identification of the area as wildlife reserve it was fenced and properly protected from livestock grazing, within a few years the recovery of the vegetation increased the chances of re-introduction of several species in the reserve as compared to areas outside the Reserve, which was overgrazed and disturbed. The local community was taken in confidence during the process and Saudi Wildlife Authority got full support both from civil society and the Government for the re-introduction of native wildlife. Arabian gazelles were obtained from King Khalid Wildlife Research Center (KKWRC). All the translocated gazelles were born in captivity at KKWRC.

Implementation: Arabian gazelles were captured just before dark and put in individual crates constructed of plywood and measuring 1.0 m x 0.36 m x 0.90 m. Crates could be opened from both ends and had 30 - 40 ventilation holes of 1 cm diameter. Animals were transported the 800 km to Mahazat at night by truck. Upon arrival at the Reserve the gazelles were placed in a quarantine enclosures (500 m x 500 m) and features to those at the KKWRC. Shade, food and water were provided in enclosure. Between March 2011 and January 2012, two groups of animals were released from the pre-release enclosure into wild when the vegetation condition was favorable.

All animals were softly released by opening gates of pre-release enclosure and animals were allowed to leave of their own, while water and alfalfa was provided outside of the enclosure for three weeks. All animals, which were radio-tagged were monitored on daily basis by ground telemetry and at least once a week by aerial telemetry using Maule aircraft and date, time, location, activity, interaction with sand gazelles, habitat and group compositions were observed.

<u>First Release</u>: In 2011 the first group of 17 (4 males:13 females) Arabian gazelles was transferred from KKWRC to Mahazat on 14th March 2011 by road. Age of gazelles was between 2 - 6 years old and ranged between 3 months old calf to 10 years old female. Radio-collars were secured to each individual with tag numbers. One female died on 19th March 2011 in release pen before release. On April 8th, 2011 two female Arabian gazelles were released directly from boxes by His Highness Prince Bandar bin Saud bin Mohammed Al Saud (SWA President). Remaining 14 gazelles (4 males:10 females) kept in pre-release enclosure were released softly by opening gate of enclosure.

<u>2nd Release</u>**:** In 2012 the second group of 23 (8 males:15 females) gazelles was transferred from KKWRC to Mahazat on 12th February 2012. Age structure of this group received is mostly 2 - 4 years old and ranged between 1 to 5 years old animals. One female gazelle was recorded dead in the pre-release enclosure before release on 21st February 2012. This group of 22 Arabian gazelles was softy released by just opening the enclosure gate on 6th March 2012.

All animals were tested for tuberculosis, vaccinated against rabies, foot and mouth disease, rinderpest, and pasteurellosis, marked with either eartags, marker collars, or radio transmitters, and placed in quarantine pens for a few months and soft released by opening the gate of the enclosure.

Post-release monitoring: In summer of 2011 and 2012, when the vegetation mostly dried off, a total of eight Arabian gazelles were recorded dead, mostly just after the release from the first release between May and November 2011. Five gazelles (1 male:4 females) went missing due to radio-collar failure and one radio collar fell of one female. These animals were not recorded again till date.


Arabian gazelles in Mahazat © M. Z. Islam

Only one female was found dead on 31st March 2012 among the second released group of 22 animals. Mortalities were controlled by further improving the release method, by releasing them in winter months not as 2011 and also by decreasing stress on animals during the second release. Another factor for successful release was the fact that the Reserve received good rainfall and that made the reserve green. Post release dispersal of Arabian gazelles have been recorded from the intensive monitoring programs. After the release the productivity of wild gazelles was high and after one year of release, the gazelles started breeding. Five radio-tagged females gave birth to one calf after one year of release and other females would produce calves too.

Breeding records of the gazelles: The first wild born Arabian gazelle calf was recorded in Mahazat on 28th August 2012 near the fence. This calf was almost one month old when recorded with the group. Three other females delivered one each by the end of September 2012. The offspring show more adaptability to the wild than to their captive-bred parents and other females were also recorded pregnant. The present population of Arabian gazelle in Mahazat Reserve is between 30 - 35 (exactly 29: 11 males:14 females:4 juveniles) animals are monitored on a regular basis. Studies related to its habitat use, feeding ecology, range and space use, and group composition are been carried out in Mahazat.

Major difficulties faced

- Maintain long-term regular monitoring.
- Lack of skills for mass capture techniques for Arabian gazelles.
- No study on the genetic diversity of gazelle in released sites has been done recently.

Major lessons learned

- When wide-ranging species are confined to restricted areas, even if such areas are large, it is essential that an effective population management plan is in place BEFORE any re-introduction is carried out and that the plan is properly implemented. If this is not done, large-scale mortalities will occur.

- Prior to any transplantation, range conditions in the release area have to be improved and the area protected from livestock exploitation. Once pasture conditions show adequate signs of improvement and the site is adequately protected, re-introduction of the animals can be contemplated.
- The time of release should coincide with suitable vegetation conditions.
- Keeping the animals in pre-release enclosures within the re-introduction site to get them acclimatized to the natural environment and provide minimal amount of food and water.
- Regulate tourism in re-introduction areas as this can lead to increased habitat degradation.
- A public-awareness program should in place to inform citizens of the biological and historic significance of the Arabian gazelle in the society.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:
- The Arabian gazelle was locally extinct in the south-western Saudi Arabia and now we have breeding populations through the captive-breeding and re-introduction programs.
- The population of Arabian gazelle withstood the drought without further supplemental re-introduction support.

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The restoration of elk in Ontario, Canada, 1998 - 2012: research and management implications

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Introduction

Elk (*Cervus elaphus*) or wapiti were historically distributed across much of North America. However, during the 1800s they were extirpated from much of their range, including Ontario, Canada, presumably due to unregulated harvesting, changes to habitat associated with settlement, and extermination by landowners due to elk competition with livestock and damage to crops and property. Numerous restoration projects were initiated across North America during the early 1900s in an effort to re-build elk populations. One restoration project in Ontario during the 1930s was successful at restoring elk to the province. However, an extinction order was issued during the 1940s and 1950s due to the incorrect belief by provincial officials that elk were spreading giant liver flukes (*Fascioloides magna*) to cattle (Rosatte *et al.*, 2007). As a result, elk were legally hunted in Ontario until 1979. A few of those elk survived and during the mid-1990s, two small herds of about 50 elk (offspring of surviving elk) still existed in the Burwash area of central Ontario. During 1998 to 2001, 443 elk acquired from Alberta, Canada, were released in 4 areas of Ontario in an attempt to bolster the Burwash area herds and restore elk to other areas of Ontario (Rosatte *et al.*, 2007). Research projects were implemented at all release sites to provide data for the effective management of elk in Ontario and in 2010 a provincial Elk Management Plan was implemented.

Cow, Calf and bull elk © R. Rosatte

Goals

- Goal 1: To restore a species that had been extirpated from Ontario, Canada during the 1800s.
- Goal 2: To determine the environmental impact of restoring elk in Ontario.
- Goal 3: To implement research and monitoring programs that would provide data for decision making regarding the management of elk in Ontario.

- Goal 4: To provide social and economic benefits e.g. recreational opportunities such as harvesting and viewing, for the residents of Ontario.
- Goal 5: To enhance biodiversity in Ontario.

Success Indicators

- Indicator 1: Sustainable elk populations in Ontario.
- Indicator 2: Mortality of elk is low and elk productivity and calf survival is high.
- Indicator 3: Elk interactions with humans are minimal.
- Indicator 4: Elk competition with other ungulates such as deer is minimal.
- Indicator 5: Elk damage to the environment is minimal (including disease/ parasite spread).

Project Summary

Feasibility: Historically, elk were found in some areas of Ontario, suggesting that certain habitats in the province could support elk populations. During the mid to late 1990s, the Plan for the Restoration of Elk in Ontario was drafted and approved by the Ontario Ministry of Natural Resources (OMNR). The plan identified six broad geographic areas of the province where elk could be potentially restored and the recommendation was that at least 200 elk should be released in each area selected as a release site. A habitat supply model was used to determine which areas of the province could support elk populations. Each area was ranked according to elk ecological variables, mortality risk, potential for elk interaction with humans, and logistics of restoring elk in that area (Bellhouse & Broadfoot, 1998).

Implementation: Ontario's elk restoration program was a multi-partnered collaboration with members from provincial and federal governments, colleges and universities, private organizations, and volunteers. Elk Restoration Advisory and Technical Committees were established to coordinate the overall implementation and delivery of the Ontario elk restoration program. In addition, Local Implementation Committees (LIC's) were established in the release areas to oversee release site logistics. Elk were captured and processed (tested for diseases such as brucellosis and bovine tuberculosis, administered anti-parasitic agents, sexed and aged, and were fitted with radio collars and ear-tags) at Elk Island National Park, Alberta (Rosatte *et al.*, 2007). Elk were not accepted for shipment to Ontario unless they were declared disease-free by Canadian Food Inspection Agency (CFIA) veterinarians. Elk were shipped to Ontario via Rocky Mountain Elk Foundation (RMEF) trailers or commercial stock trailers. Upon arrival in Ontario elk were placed into holding pens to recover from the 1 to 2 day journey. A total of 443 elk were released after a variable holding period at four areas in Ontario: the Lake of the Woods (LOW) area near Kenora (104 elk), the Lake Huron North Shore (LHNS) area near Blind River (47 elk), the Nipissing/ French River (NFR) area near Sudbury (172 elk), and the Bancroft North Hastings (BNH) area near Bancroft, Ontario (120 elk) (Rosatte *et al.*, 2007). The Ontario elk restoration plan had identified six potential areas in Ontario that could receive elk. However, following a risk assessment, a moratorium on the shipment and release of additional elk in Ontario was implemented in 2001 due to the perceived


Rick Rosatte with drugged elk © J. Neuhold

risk of importing Chronic Wasting Disease (CWD) into the province. Ontario currently (January 2013) remains free of CWD.

Post-release monitoring: Following the release of the elk in Ontario during 1998 to 2001, program staff, volunteers, college and university students, monitored their movements and survival using radio-telemetry as most of the elk were collared. Twelve graduate student programs were also initiated at four Ontario universities during 1998 to 2012 to study the dynamics of the elk herds at the four elk release areas. From 1998 to 2004, mortality of released elk was about 41%. Causes of mortality included predation by wolves (primarily in the NFR and LOW release areas), illegal shooting, collisions with vehicles, infections, and emaciation (Rosatte *et al.*, 2007). However, mortality has since declined as elk became acclimated to their new home in Ontario. It was also found that in some of the release areas the length of time the elk were kept in pens prior to release had an effect on their dispersal distance. In fact, extended holding periods (up to 4 months) promoted philopatry (Ryckman *et al.*, 2010). Another research study determined that there was a moderate amount of dietary overlap between elk and resident white-tailed deer (*Odocoileus virginianus*) in the BNH release area (Jenkins *et al.*, 2007). McIntosh *et al.* (2007) found that about 59% of elk sampled in the BNH release area were infected with meningeal worm (*Parelaphostrongylus tenuis*); however, the full impact of that parasite on elk survival in Ontario has yet to be determined (Bellhouse & Rosatte, 2005). Research on the dynamics of elk populations in the four release areas in Ontario continues to-date (2013).

A provincial Elk Management Plan was developed with public input and implemented by OMNR in 2010. This plan aligns with the Cervid Ecological Framework which is in place to manage cervids (moose, elk, deer, caribou) at the ecosystem or landscape level in Ontario. Monitoring of elk to date has revealed that the BNH and LHNS elk populations are doing extremely well. In fact, a hunt was initiated in the BNH area during 2011 to assist with managing the herds at a desired population objective (400 to 600 elk in the BNH core release area which is about 2,500 km^2) and provide recreational opportunities to Ontarians. The elk hunt may also help to reduce human/elk conflicts especially on agricultural lands in the BNH area. Elk in the NFR area suffered high mortality during the initial stages of restoration due to wolf predation and a variety of other mortality factors. That population has struggled but currently (2013) appears to be recovering.

However, the LOW elk population appears to be struggling due to a variety of factors. The provincial elk population estimate for the four core elk ranges in Ontario during 2012 was 648 to 916 elk.

Major difficulties faced

- Transport of elk in trailers for 24 to 58 hours continuous driving during winter conditions was stressful to elk as well as the truck drivers.
- Elk escaped from one of the release area pens (BNH) on day 1 of the holding period and dispersed over a 27,000 km^2 area (Rosatte *et al.*, 2007).
- Elk mortality was initially high following release due to a number of factors including wolf predation, drowning, collision with vehicles, illegal shooting, infections and emaciation.
- Had to address concerns raised by naturalist groups and hunters regarding the environmental impacts of releasing elk in Ontario.
- Had to assess the risk of Chronic Wasting Disease being imported into Ontario via shipment of elk from Alberta.
- Conflicts with area farmers has become an on-going challenge.
- Divisive issue as to whether elk should have access to supplemental food sources.
- Acknowledged the potential for restored elk to interbreed with escaped captive elk, red deer, and hybrids in Ontario and attempts have been made to remove escaped captive animals from the landscape.
- Determining if a re-introduction is successful and deciding when to cease efforts to sustain a herd/population that is facing continued decline.

Major lessons learned

- Elk should be placed in holding pens prior to release for 1 to 4 months to allow them to recover from the stress of relocation, acclimatize to their new environment, and promote fidelity to the release site area.
- Prime animals should be selected for restoration to maximize productivity during the initial stages of restoration.
- Elk released in areas of high predator (e.g. wolves) density will in all likelihood experience high mortality during the initial stages of restoration.
- Hunters need to be educated to be certain of their target when elk are released into areas

Helicopter preparing to capture elk for radio collaring near Bancroft, Ontario © Rick Rosatte

where there are hunting seasons for species such deer and moose, otherwise elk mortality due to illegal shooting will likely be high.
- Human/elk conflicts will occur in areas containing agricultural croplands, and co-operative work is needed to develop effective tools for minimizing conflicts.
- If elk are restored to an area, populations need to be monitored and a comprehensive elk management plan is imperative to deal with disease/ parasite management as well as human conflict issues.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:
- Dispersion of elk away from the release site areas resulted in some animals not contributing to productivity.
- Wolf predation combined with other mortality factors has suppressed population increase during some years in the Lake of the Woods and Nipissing/French River elk release sites.
- Little predation of elk combined with high survival and productivity has resulted in significant population increases in the Bancroft North Hastings and Lake Huron/North Shore elk release areas.
- During the initial stages of the restoration program (1998 - 2004), illegal shooting accounted for 25 known elk mortalities.
- There was a moderate level of dietary overlap between elk and resident white-tailed deer.
- Some undesirable interactions with humans e.g. in some areas, elk/human conflicts have occurred in the vicinity of agricultural lands.
- Overall, the program was a success due to the collaboration of nearly 20 organizations, including provincial and federal governments, universities, private organizations, and volunteers.
- A provincial Elk Management Plan was implemented to provide direction to manage elk in Ontario.
- Elk research and monitoring programs provided input for elk management decisions.

Acknowledgments: The Ontario elk restoration and research program was a collaborative effort involving the following organizations: the Ontario Ministry of Natural Resources, the Ontario Federation of Anglers and Hunters, the Canadian Food Inspection Agency, Parks Canada, Elk Island National Park, Cambrian College, Trent University, Lakehead University, Laurentian University, the University of Guelph, Sault College, French River Resorts Association, the Rocky Mountain Elk Foundation, the Ontario Fur Managers Federation, Safari Club International (Ontario and Ottawa chapters), as well as the numerous volunteers associated with each elk release area. The manuscript was reviewed by the following OMNR staff: Dr. C. J. Davies, D. Stetson, V. Ewing, E. MacDonald, J. Holder and G. Lucking.

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Re-introduction of bison to the Wind River Ranch in northern New Mexico and Native American lands in the western USA

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Introduction

Bison (*Bison bison*) were catalogued as Near Threatened by the World Conservation Union (IUCN) in 2012. Bison, a mega-herbivore in North America, numbered around 30,000,000 when Euroamericans arrived. During the 1870s and 1880s, all but about 1,000 were slaughtered in a conscious attempt to remove the primary resource for Plains Indians, disrupt Indian lifestyle and culture, and clear the way for settlers and cattle. The slaughter also removed a species from the ecosystem that played a key role in the maintenance of healthy grasslands. Efforts to conserve the few remaining bison began around 1900. The return of bison can reestablish hope and culture to tribes, as well as re-establish health to a badly abused grassland ecosystem. Even though there are presently 400,000 bison in North America, 97% of them are managed for meat, not for conservation and ecological function. Since 1992, the Inter-Tribal Buffalo Council (ITBC) has restored 15,000 bison to tribal lands, yet the bison is still the only major ungulate that has not recovered following the wildlife declines of the 19th century. The Wind River Ranch (WRR) and the ITBC currently seek to expand research on the ecological role of bison in grassland health.

Bull bison and cow in typical habitat

Goals

- Goal 1: Establish a conservation herd at a level that will allow bison to perform their ecological function on the grassland.
- Goal 2: Research the ecological function of bison in grassland health.
- Goal 3: Analyze the genetics and lineage of the animals in the herd.
- Goal 4: Cooperate with the ITBC on bison research, management, and cultural issues.

Success Indicators


Adult bison with calf © Jim Stone

- Indicator 1: Maintain between 40 and 60 adult equivalents in the bison herd, adjusted according to conditions on the ranch. The herd is half owned by the Jicarilla Apache and half by the WRRF.
- Indicator 2: Performing research on the role of bison as the native mega-herbivore in a functioning grassland. We have had one pilot study, one MS project that is completed, and one 5 year study that is in its first year.
- Indicator 3: The genetic research is ongoing. It has identified one completely new lineage, and another lineage that comes from the Yellowstone herd.

Project Summary

We began managing the WRR in January of 2005, following several years of drought and grazing by a herd of horses. We rested the grass until 2007, when we started grazing nine bison owned by the Jicarilla Apache Office of Cultural Affairs (JAOCA). Our intention was to give their herd a head start until they could get permission to graze bison on tribal lands. In 2008, the ITBC donated 35 more bison to the JAOCA herd at the WRR. The JAOCA donated 3 females and one male from that group to the WRR. In 2009 and 2010, WRR grazed a dozen bison from the Picuris Pueblo. WRR then bought those bison from the Picuris tribe. Presently, there are 68 individuals of various ages from calves to adults in the herd, with half owned by the WRR and half owned by the JAOCA. The bison respect our 1.2 m high barbed wire external fence. We have removed internal fences except for a trap when we want to put bison in the corral.

The WRR and ITBC cooperate on this herd of bison, with the ITBC paying the salary of a bison caretaker during 2011. We manage the bison as a conservation herd, and to assist tribes with bison. Because we want to investigate the role of bison in grassland health, we maintain a number of bison that is large enough to have ecological impact, but not so large as to degrade grasses. When we have excess animals, they are sold to other tribes, sold for meat, or enter the JAOCA free meat program. We periodically monitor the grass condition, and each fall we assess the amount of grass we have for winter grazing. We do this by mapping the grass conditions around the ranch, and estimating the pounds of grass per acre in those various areas. We convert the various ages of bison into adult equivalents of 450 kg and assume that each adult equivalent will eat about 9 kg of grass per day (2% of body-weight). We then calculate how many bison can live

Bison in Yellowstone during winter © Jim Stone

on the grass for the next nine months. As a conservation herd, we are trying to move closer to a 50:50 sex ratio, although at present we are biased toward females. Genetic analyses, both mitochondrial DNA and nuclear DNA, indicated that these bison represent the Yellowstone lineage as well as several that are in a lineage not previously described.

A pilot study and a Master's Thesis have both shown that bison break piñon, juniper, and yucca that advance onto the grassland. Similar to elephants in Africa, bison probably played a role in halting the transition from grassland to savannah to woodland, a transition that has degraded millions of acres of grassland in the western U.S. as well as causing arroyo formation. Arroyos lower the water table and reduce the productivity of the surrounding grassland. An ongoing study at WRR by the Denver Zoo is investigating flora and fauna associated with bison compared to cattle. WRR and ITBC are planning to develop more studies of the bison's role in grassland function.

The ITBC is composed of 57 tribes, and the organization's members currently have a population of 15,000 bison on 51 reservations in 19 states. Bison historically had a wide range in North America so the number of tribes interested in restoring bison for cultural reasons is varied. Of great importance to tribes is regional research on how bison restore lands that were grazed by cattle. This allows tribes to determine what changes and progressions can be seen in their own restoration efforts. The Southwest is a unique ecosystem, and that makes it hard for Southwestern tribes to extrapolate from previous efforts by northern tribes. The documented results of the WRR/ITBC restoration will be of the utmost importance to the regional tribes, allowing them to develop management principles that are science based.

Major difficulties faced

- Finding grant money to do the research.
- The WRR has only about 4,600 acres and is not large enough to have a large bison herd.
- They reproduce well and are long-lived, so we need to watch numbers, but the ITBC has been able to help move excess animals to other tribes.

Major lessons learned

- The role of bison in preventing the transition of grassland to savannah to woodland.
- That bison are much easier to work with than many people say.
- Cooperation between like-minded groups is important for long-term conservation of bison.

Success of project

Highly Successful	Successful	Partially Successful	Failure
√			

Reason(s) for success/failure:

- The fecundity of bison and the cooperation between the ITBC and the WRRF.

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Restoration of wisent population within the Carpathian eco-region, Europe

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Introduction

The wisent or European bison (*Bison bonasus* L.), fairly common in the Carpathian eco-region (a mountain range of about 210,000 km^2) in Medieval Ages, finally disappeared there by late 18th century, due to overhunting and gradual habitat loss. First attempts for its re-introduction date back to the 1960s (Poland and Ukraine). The restitution project involving five countries (Poland, Slovakia, Ukraine, Romania and Hungary) was initiated in the late 1990s. By 2012, there are 6 free ranging wisent populations counting together over 350 individuals and seven breeding enclosures (100 animals). Planned activities concentrated on increment of wisent numbers, the extension of their range and an improvement of the genetic pool through prescribed supplementation with selected individuals from captivity. The species has IUCN status Vulnerable. It is divided into two genetic lines, Lowland with VU status and Lowland-Caucasian classified as Endangered (EN) because of decreased population size. The species is listed in Appendix III of the Bern Convention, and on Annexes II* and IV of the EU Habitats and Species Directive. In Polish Red List the species has category EN and most countries in which the species occurs have national management plans. The European Bison Conservation Centre established in last years is responsible for coordination and information exchange.

Wisent bull in Romania © Kajetan Perzanowski

Goals

- Goal 1: The establishment of a viable meta-population of the species in the Carpathian eco-region.
- Goal 2: An improvement of present genetic structure of Carpathian herds.
- Goal 3: An extension of the present range of the species in the eco-region.

- Goal 4: An increase of a number of free-ranging herds.
- Goal 5: An introduction of a routine monitoring of all free ranging herds in the region.

Release at Bieszczady National Park
© Kajetan Perzanowski

Success Indicators

- Indicator 1: Reaching an effective population number over 500 individuals.
- Indicator 2: An increase of underrepresented founders in the gene pool of this population.
- Indicator 3: An establishment of free ranging herds in Romania and Slovakia.
- Indicator 4: Spontaneous migrations of animals outside of herds' home ranges.
- Indicator 5: Acceptance of free ranging wisents by local communities.

Project Summary

Feasibility: The Carpathians are the largest and most important linkage for wildlife between the south-eastern and central part of the continent. This mountain chain remains mostly forested (from about 30% in Hungary up to over 60% in Romania), including the largest in Europe stretch of natural mountain forest dominated by fir and beech. It is also a mainstay of a majority of native large mammals including almost all European large predators: brown bears, wolves and lynx. The wisent, extirpated from the region some 200 years ago, was the last surviving species of large grazers, contributing in the past to the maintenance of grassland communities and forest mosaic. Gradual encroachment of settlements into mountain valleys, and the development of livestock based local economy have led to the fragmentation and the loss of a large part of natural habitats. Economic and political changes after the World War Two did not facilitate a cooperation in the field of nature conservation within the region. A majority of forests, and a considerable part of cultivated land became state controlled and subject to central planning. First attempts to bring back wisents to the region were undertaken independently some 50 years ago in Poland and Ukraine, but internationally coordinated project became possible only by the end of the 1990s of the 20th century.

Implementation: The project was initially based upon already existing free ranging herds (two in Poland and two in Ukraine) but gradually it was extended to Slovakia, Romania, and Hungary. According to the guidelines determined by the European strategy for the conservation of the species (Pucek *et al.*, 2004), concerned with separate maintenance of two genetic lines (Lowland and Lowland

Loading of wisents in Ukraine

© Lukasz Polawski

–Caucasian), since first wisent released there belonged to the latter, further introductions followed the same rule. The source of new animals are genetically selected from various breeding centers of Europe. Because of exceptionally high levels of inbreeding within the species, the main criterion for their choice is the genetic distance and founder representation. The genetic evaluation is based on pedigree analysis as well as on DNA genotyping, mainly microsatellites. In the beginning of the project, all involved countries did not belong to the EU, so an import of animals was legally and logistically quite complicated. Now, only Ukraine remains outside of the EU so large part of necessary arrangements and paperwork is much easier. A serious problem remain however health related issues, since as Bovines, wisents may transmit various diseases dangerous to the livestock including foot and mouth disease, brucellosis or TB. Also the legal status of this species is not the same all over the Europe, ranging from fully protected to being listed among cattle. So far in countries of western Europe, wisents are maintained only in captivity and their release to the wild is considered as highly controversial, however there are plans for such experiments in Germany, Sweden, Holland and Denmark.

Post-release monitoring: In countries where wisents enjoy the freedom, their numbers, population structure and movements are monitored either by Forest Service or national park personnel. As a rule, samples of tissue are collected from dead animals, and in the case of Polish population also seasonally samples of feces as an indicator of parasitic infestations. Since 2002, in the majority of cases, wisents released to the wild were fitted with radio-collars allowing to verify their interactions with wild animals and follow their fate. There is an exchange of information on this subject among neighboring countries (Poland, Slovakia and Ukraine), and results of monitoring are published on regular basis in commonly accessible international journals.

Major difficulties faced

- Problems connected with transfer of animals between countries (health status) and between EU and non EU countries (legal status).
- Obtaining a consensus with local stakeholders.
- Lack of stable financial support for established free ranging herds.

- Uncontrolled losses of animals due to poaching in Ukraine.

Major lessons learned

- The project was carried out in countries of various economic conditions and different legislations regarding nature conservation so every time a different approach was required to tackle any arising issues.
- A key for the success of newly established herds is an acceptance of the presence of introduced animals by local communities.
- There is a threshold regarding the size of a population (about 40 animals), below which its numbers grow very slowly and the population remains vulnerable to extinction.
- Free ranging herds should be monitored on long term basis, including: population census, spatial distribution, and mortality causes.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- A number of people dedicated to the conservation of this species.
- High level of social acceptance for the species in countries of the region.
- Fairly well maintained natural and semi-natural habitats.
- Well-developed methods for captive breeding of the species.
- Broad access to captive animals suitable for introduction to the wild.

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Re-introduction of wood bison in Russia

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Introduction

In the Pleistocene and early Holocene, bison inhabited almost the entire territory of Russia. In historic times, its range began to dwindle and became fragmented (Lorenzen *et al.*, 2011). The main reasons for this were climate change and persecution by humans. Up till the 7th - 10th century, the bison lived in the trans-Baikal and Baikal regions (Ermolova, 1978; Vereschagin & Baryshnikov, 1985). To the south of Yakutia the bison populations merged about 2,000 years ago (Lazarev *et al.*, 1998), and to the northeast of Siberia about 3,000 years ago (Flerov, 1977; Sipko 2009). According to K.K. Flerova (1977 & 1979) bison in eastern Siberia are almost identical to modern wood bison (*Bison b. athabasckae*) now found in Canada and these animals are listed on CITES. The high degree of similarity in the ecosystems of both Canada and Russian Siberia has identified an interest in the return of wood bison. For this purpose, two sites were selected on the territory of Yakutia. An important element in the idea of this project was the need to create the most isolated population of wood bison, giving assurance that the animal form will be retained, in case of any mishap threatens them in Canada.

Goals

- Goal 1: Establish a breeding center for these animals, to preserve the gene pool of this species. The isolation provided by the great distance guarantees the obstacles from simultaneous occurrence of any threat to these animals.
- Goal 2: Return species previously present in the ecosystem of Siberia.
- Goal 3: Raise the productivity of ecosystems.

Wood bison (*Bison b. athabasckae*)

Success Indicators

- Indicator 1: Achieve sustainable regeneration of herds and get the offspring of calves born in Siberia.
- Indicator 2: Assemble on the territory of Yakutia a genetically diverse population.

- Indicator 3: Get practical examples of successful bison re-introduction in the wild of Yakutia.

Project Summary

The project for the re-introduction of wood bison in Russia was implemented by the importation of 90 bison from Elk Island National Park in Canada. The animals were brought in groups of 30 individuals for several years. The history of the wood bison started in 1965 when 23 individuals were delivered which became the population founders. Also we assumed that the imported bison had all the genetic polymorphism. The imported bison have successfully adapted to the local climate and the low winter temperatures. This is an important consideration, since in December - January the average temperature is $-40^{0}C$ and the temperature is often below $-50^{0}C$. In the summer season there is a large number of mosquitoes and heat, which is a problem for ungulates.

Wood bison were placed in two breeding centers, located at a distance of 200 km from each other and in a place with a low density of domestic animals. This is important to ensure the biosecurity of the bison. The bison began to successfully reproduce, and their population dynamics can be seen in table 1. Supplementary feeding is available only during the winter but a few experiments were also conducted on male bison without supplementary winter feed in the wild. These experiments were successful and the results will be integrated into more successful management of the wild bison.

Table 1. An update of bison numbers from the two breeding centers

	Year							
	2006	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**
Breeding Center, Lenskii Stolby National Park								
Imported from Canada	30	0	0	0	0	0	0	0
Birth	0	0	6	7	8	10	5	?
Death	2	2	0	0	0	0	0	0
Removed	0	0	0	6	7	8	9	5
Year end total	***28***	***26***	***32***	***33***	***34***	***36***	***32***	***27***
Breeding Center, Timpinay National Park Sinyy								
Imported from Canada	0	0	0	0	0	30	0	30
Imported from Buotoma	0	0	0	6	7	8	9	5
Birth	0	0	0	0	0	0	2	?
Death	0	0	0	0	0	2	0	0
Year end total	***0***	***0***	***0***	***6***	***13***	***49***	***60***	***95***
GRAND TOTAL	**28**	**26**	**32**	**39**	**47**	**85**	**92**	**122**

Overview of the Yakutia territory

Major difficulties faced

- This project is ongoing with support and funding from the Government of the Republic of Yakutia, which ensures its sustainable development.
- Problems of adaptation to the harsh climate of this unique region were overcome as well.
- A problem remains with the presence of bears living in the nearby national parks.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:
n/a

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Establishment of a desert bighorn sheep population to the Fra Cristobal Mountains, New Mexico, USA

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Introduction

Desert bighorn sheep (*Ovis canadensis mexicana*) were once prolific in New Mexico, occupying most arid mountain ranges in the southern part of the state. Over-hunting and disease transmission from livestock are two primary reasons for the dramatic decline in bighorn sheep numbers throughout the West during the early 1900s; desert bighorn sheep in New Mexico responded similarly. In 1980, the desert bighorn sheep was placed on New Mexico's endangered species list (Goldstein & Rominger, 2003). That led to a concerted recovery effort that included re-introduction projects to establish populations throughout southern New Mexico, including the Fra Cristobal Mountains. Efforts to establish desert bighorn to the Fra Cristobal Mountains in southwestern New Mexico began in 1995 as a collaboration between the New Mexico Department of Game and Fish (NMDGF) and the Armendaris Ranch (owned by conservationist and philanthropist Ted Turner) (Goldstein and Rominger 2003).

That year 13 rams and 24 ewes were translocated from the fenced bighorn refuge managed by NMDGF to the Fra Cristobal Mountains. The releases marked the beginning of the only effort in New Mexico to establish the species on private land. In 1997 an additional seven radio-collared rams were released in the Fra Cristobals.

Desert bighorn sheep at Eagle Rock

Goals

- Goal 1: Restore a self-sustaining population of desert bighorn sheep to the Fra Cristobal Mountains that is large enough to persist over a long period of time (≥100 years) with little or no human intervention.
- Goal 2: Manage the restored herd to accommodate a

recreational fee hunt that generates funds to offset the cost of the restoration project.

- Goal 3: Publish important findings from the restoration project to advance the science of restoration ecology.

Desert bighorn sheep translocation

Success Indicators

- Indicator 1: Presence of a collaborative working relationship with the state of New Mexico in which roles and responsibilities are defined.
- Indicator 2: Re-introduction of an adequate number of desert bighorn sheep to catalyze population establishment.
- Indicator 3: Establishment of a monitoring and research framework that is adequate to support adaptive management of the restoration project to maximize the probability of success.

Project Summary

In 1997 the Turner Endangered Species Fund (TESF) took a leadership role in the desert bighorn restoration project. In the ensuing 15 years TESF worked collaboratively with the Armendaris Ranch and NMDGF to monitor bighorns and cougars and their interactions on a near daily basis. TESF also spearheaded several research projects that aimed to advance sheep population establishment. Diseases, acquired primarily from domestic sheep, had for more than a century dominated the documented and suspected causes of extinction of wild bighorn from causes other than hunting (Singer *et al.*, 2001). But the mid-1990s it became known that the proximate cause of most mortalities leading to extinction of small populations of desert bighorn turned out to be not disease but predation. mountain lions (*Puma concolor*), or cougars, emerged as the main predator (Goldstein & Rominger, 2003).

Release in the Fra Cristobal Mountains in Armendaris Ranch, New Mexico

In New Mexico cougar predation dominated among the factors responsible for the demise or poor performance of five desert bighorn populations that the state had actively managed from 1992 through 2002. In several other southwestern locations, biologists also recognize cougar predation as a major impediment to bighorn herd re-establishment and replenishment (Rominger *et al.*, 2004). Six of the seven radio-collared rams released in the Fra Cristobal Mountains in 1997 were killed by cougars within 18 months. Curiously, few bighorn biologists prior to the 1980s had thought cougar predation very important as a desert bighorn mortality factor (Geist, 1971).

While the circumstances that led to cougar predation becoming an important factor affecting the persistence of desert bighorn sheep populations are not well understood, the importance of the factor is undeniable. From this simple fact we designed a restoration scheme that focused on cougar monitoring and removal, using lethal means, to minimize cougar predation of sheep that inhabited the Fra Cristobal Mountains. The intensity of cougar control was inversely related to the perceived threat represented by the animal(s) in question. Adult females with dependent young that restricted movements to the Fra Cristobal Mountains represented a pronounced threat to sheep and were immediately targeted for removal. In contrast, wide-ranging adult males were identified as a modest threat and only targeted for removal if they restricted movements to areas frequented by sheep.

From 1997 through mid-year 2011, we used telemetric monitoring and remote camera “traps” to document cougar use of the Fra Cristobals and to instruct removal actions. During this period we removed 34 cougars from the area. Concurrent with this by May 2011 the sheep population had grown to include 200 to 220 animals and had catalyzed (through emigration) a second population on the nearby Caballos Mountains (about 20 km south) that included 65 to 75 sheep. This “meta-population” of 265 to 295 sheep was the largest in New Mexico and included over 40% of all sheep in the desert bighorn sheep in the state. The Fra Cristobals/Caballo mountains meta-population was the principal reason that the New Mexico State Game Commission removed the species from the state list of imperiled species in November 2011.

Just prior to delisting TESF recognized the successful restoration of desert bighorn sheep by approving the removal of 16 ewes from the Fra Cristobal Mountains for conservation purposes. On 30th October 2011 these animals were captured and translocated to suitable habitat elsewhere in New Mexico to advance the species' security. This management action represented the first time in history that desert sheep have been restored to private property and managed so successfully that the herd grew to sufficient size to serve as a "donor population" to support range-wide conservation efforts. Starting in 2012 the Fra Cristobal Mountains desert bighorn sheep population became the target of a recreational harvest of trophy rams.

Given that only a small percentage of rams breed, an annual harvest of a few "trophy" rams can be sustained without affecting population vigor and persistence. During the fall of 2012 six bighorn rams, including five that qualified for the Boone & Crockett record book (www.boone-crockett.org), were harvested. Three of these animals were harvested according to permits issued by NMDGF to the Armendaris Ranch. The ranch was able to sell these permits to hunters for US$ 165,000. From this total the Armendaris Ranch donated US$ 55,000 to offset the cost of operating the Beau Turner Youth Conservation Center in Florida.

Cougar management continued in 2012 as well. By mid-December of that year ranch personnel working in tandem with NMDGF had removed five lions from the mountain, including three males and two females. Work plans for 2013 and beyond include continued monitoring and management of cougars to minimize predation on sheep along with recreational, high dollar hunts of trophy rams.

Major difficulties faced

- Blending management actions and research efforts in a manner that informed adaptive management while not compromising the growth capacity of the nascent population of desert bighorn sheep.
- Maintaining a field crew capable of successfully carrying out the chronic monitoring under difficult field conditions to ensure completion of management actions and research efforts necessary to ensure the restoration of a viable population of desert bighorn sheep.
- Maintaining collegial and effective relations

Cougar photographed by remote camera

between the state of New Mexico and the owner of the land to which the sheep were released.
- Balancing the tension created by establishing lethal control of cougars as a requisite to desert bighorn sheep restoration.

Major lessons learned

- Predator control, in this case involving cougars, to promote the growth of a prey population, in this case involving endangered desert bighorn sheep, is controversial and has notable potential to create tension within a restoration team and between the general public and the restoration team.
- All members of the restoration team, from field biologists to senior administrators, need to be forever mindful of the difficulty of overcoming the many forces that operate against endangered species restoration efforts including environmental, logistical, fiscal, intellectual, and socio-political.
- It is difficult to blend monitoring activities and research efforts in a manner that does not compromise the project's principal aim - restoring a viable population. This potential to compromise success can create tension within a restoration team over the proper role of research in an endangered species restoration project.

Success of project

Highly Successful	Successful	Partially Successful	Failure
√			

Reason(s) for success/failure:

- Implementation of a systematic approach to minimize if not completely eliminate cougar predation of desert bighorn sheep.
- An effective collaborative partnership between private non-governmental conservation organizations and the New Mexico Department of Game and Fish.
- A clear understanding that restoration of a viable population of desert bighorn sheep would require an extended period of time over which chronic, near daily fieldwork would be needed to provide current information about sheep and cougars to inform management actions.
- A private landowner deeply committed to endangered species restoration.

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Twenty years of captive breeding and re-introduction of the eastern barred bandicoot in Victoria, Australia

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Introduction

The eastern barred bandicoot (EBB; *Perameles gunnii*) is a small (<1kg) nocturnal marsupial. It is solitary, short-lived (2 - 3 years) and highly fecund (i.e. up to 5 litters a year; average litter size 2 - 3). EBBs are omnivorous and opportunistically exploit a wide variety of invertebrates and some plant matter. The EBB was once found across the Basalt Plains of Victoria, Australia. The original vegetation was perennial tussock grassland with areas of grassy woodland, but 99.9% of this habitat has been destroyed or modified for agriculture. The introduction of the red fox *(Vulpes vulpes)* caused a catastrophic decline in the EBB population and by 1989 only 150 - 300 individuals remained. The last wild population was declared extinct in 2008. The first EBB management plan was developed in 1989 and re-introduction has been attempted at eight sites with varying success (Hill *et al.*, 2010). Only two of these sites currently support extant populations. The Victorian EBB is listed as Endangered on the Commonwealth

Eastern barred bandicoot

Environment Protection and Biodiversity Conservation Act 1999 *and* Critically Endangered on the Advisory List of Threatened Vertebrate Fauna in Victoria 2007.

Typical habitat of the eastern barred bandicoot

Goals

- Goal 1: The long-term objective of the eastern barred bandicoot recovery program is to minimise the probability of extinction by establishing at least two self-sustaining re-introduced populations which total a minimum of 2,500 individuals.
- Goal 2: Establish self-sustaining re-introduced populations totalling at least 1,000 individuals, within 5 years (2009 to 2014).
- Goal 3: Prevent further loss of genetic diversity by managing captive and released populations as one meta-population.
- Goal 4: Maintain a viable insurance population in captivity.
- Goal 5: Maintain and enhance community and institutional support.

Success Indicators

- Indicator 1: Self-sustaining populations established at a minimum of two large re-introduction sites.
- Indicator 2: At least 1,000 EBBs in “wild” populations by 2014, and 2,500 EBBs in “wild” populations by 2024.
- Indicator 3: Less than 5% loss of genetic diversity of the species from the inception of the captive breeding program,
- Indicator 4: Multiple captive facilities for holding and breeding EBBs to spread risk and provide a source of animals for re-introduction.
- Indicator 5: Active recovery team with effective working partnerships between government, zoos and university researchers.

Project Summary

Feasibility: The EBB was formerly widespread in Victoria, occupying a total range of three million hectares. Following European settlement and the introduction of feral predators, in particular the red fox, bandicoots suffered a significant decline. The EBB requires structurally complex habitats with dense cover for nesting, adjacent to open areas suitable for feeding, but 99.9% of the preferred grassland habitat has been cleared or modified for agriculture. By 1972, the EBB was extinct throughout its mainland home range, except for a small population within a 600 ha area in Hamilton, Victoria. In 1982, the first (interim) management prescriptions and PVA were produced in an attempt to conserve the species and the final plan was released in 1989. In 1988, EBBs were considered at risk of extinction and trapping was conducted to commence a captive breeding program (Hill *et al.*, 2010).

Implementation: In 1988, ~40 EBBs were caught to start an intensive breeding program, in which 19 founders produced 54 young. The first EBB re-introduction site was Woodlands Historic Park in 1989, closely followed by Hamilton Community Parklands in 1991 (Winnard & Coulson, 2008). Both sites are surrounded by a predator barrier fence and had initial success, but both went extinct due to a combination of drought, fox incursions and overgrazing by eastern grey kangaroos *(Macropus giganteus)* and European rabbits *(Oryctolagus cuniculus)* (Winnard & Coulson, 2008). Five unfenced re-introduction sites were also established on public and private land with fox control conducted regularly. All sites failed quickly, with the exception of Mooramong, a working sheep farm. This population persisted for 17 years, but is now undetectable by trapping. Due to the devastating effect foxes have on small re-introduced populations of EBBs, the Victorian Government's Department of Sustainability and Environment Animal Ethics Committee halted all releases into areas that could not be maintained fox free. Although sensible, this limits the number of sites in which EBBs can be released and significantly increases the expense of re-introductions, as building and maintaining predator barrier fences is costly.

Currently two sites hold healthy populations of bandicoots and are considered to be at carrying capacity. Mt. Rothwell, a 400 ha fenced reserve established in 2002 currently houses the largest EBB population, but the exact population size is unknown. In 2005 the Hamilton Community Parklands predator barrier fence was upgraded, bandicoots were released in 2007 when all foxes had been removed, and EBBs are now spread throughout the reserve. The first island trial introduction commenced on fox-free French Island, Victoria, in mid-2012 and releases into fenced areas at Woodlands Historic Park and Werribee Open Range Zoo are planned to commence in late 2012. Genetic analyses of the 20-year breeding program, in which 1,078 young have been produced, show that there was a significant loss of genetic diversity within EBB populations prior to the commencement of the captive program, with slight, but steady, reduction in genetic diversity during the breeding and release program (Weeks, 2010). Investigations into techniques to maintain or improve genetic diversity and reproductive fitness, including mate choice research, are underway. Based on genetic and population analyses, a meta-population management plan with an increased number of captive breeding pairs is proposed, but further research into the success of translocated bandicoots when re-introduced into an established population is required.

Post-release monitoring: Post-release monitoring generally occurs quarterly over two nights, with cage traps set on established grids. In large or unfenced reserves, trapping is focused on the area with the most EBB activity (i.e. foraging digs and/or spotlight sightings). However, the Mt. Rothwell reserve contains other small mammals that saturate the traps and reduce EBB capture rates. Here,

Monitoring of released bandicoot

camera traps have been deployed annually to monitor the population, but animals need to be caught in order to implement the new meta-population management plan. Radio-tracking has been used to determine foraging and nesting locations, but attaching radio transmitters to this species is problematic. Bandicoots have a low tolerance for collars, and several other methods of attaching transmitters have all resulted in short attachment times (5 days – 5 weeks). Tail mounted transmitters have been the most successful method to date, but provide only short-term data (<35 days). Intraperitoneal transmitters are currently in use for a 12-month trial on French Island. Whilst they provide several months of battery life and overcome the attachment problems, their short operating range (about 50 m) limits their value. Despite these difficulties, tracking shows that bandicoots usually nest within woodland areas, changing nest location regularly, and forage at night in the open grasslands. After release into an empty reserve, male bandicoots investigate large areas before settling, whilst females tend to stay in the area of release. Trapping has shown that bandicoots are heaviest and produce the most young during the cooler wetter months, whereas in summer, the numbers trapped decreases and animals are more commonly in poor condition.

Major difficulties faced

- The introduced red fox is the major threat to EBB populations. Foxes are widespread throughout Victoria and are difficult to control, making predator barrier fences essential. Constructing and maintaining predator fences significantly increases the costs of EBB recovery and reduces the number of sites available for release.
- There is a lack of suitable habitat to establish release sites because 99.9% of native grasslands have been destroyed or modified.
- Herbivore populations can increase rapidly in fenced reserves causing significant habitat degradation by overgrazing. Kangaroo control is difficult because there is a strong protective response towards these iconic species and rabbit control can negatively affect EBBs, which have been known to occupy rabbit burrows.
- Captive breeding facilities are limited. Bandicoots can live to six years old in captivity, but do not breed past ~3 years old. Post-reproductive animals can fill enclosures required by breeding animals. Furthermore, if release sites are not available, the housing of young bred in captivity can halt future breeding.

Juvenile (joey) eastern barred bandicoot

- The presence of many other small mammal species that saturate the traps at Mt. Rothwell make monitoring and trapping EBBs difficult. New trapping techniques are required to monitor the population, to trap animals for translocation as part of the meta-population and to increase the genetic diversity of the captive breeding population.

Major lessons learned

- The Victorian EBB would be extinct without the captive-breeding program initiated by Melbourne Zoo. All recovery potential has been driven by the success of captive-breeding and release. This highlights the importance of early intervention in collecting founders to establish captive populations of threatened species.
- Foxes are the key threatening process for the EBB. Populations are unable to maintain themselves unless foxes are permanently eradicated from the area. This involves surrounding reserves by a predator barrier fence, as well as continuous fox monitoring and control. The quality of habitat is less critical to this species.
- Managing each re-introduced population as a separate entity has not been ideal. The small size of reserves increases the likelihood of stochastic events having a detrimental effect on populations and can contribute further to the loss of genetic diversity. New release sites that can sustain large populations plus managing as a single meta-population will assist in the conservation of this species.
- Successful long-term captive breeding programs for marsupials are rare. This is an example of a long-term program in which animals have not decreased in reproductive rate for more than 20 years. However, careful management of space is imperative, as breeding is restricted by lack of suitable enclosures when young cannot be released. The meta-population model for the EBB requires breeding animals to be cycled in and out of the wild every two years to maintain genetic diversity. This model is also being applied to several other threatened species.
- The EBB recovery team was restructured in 2011 and three groups were formed: the science, operational and business groups. These groups meet on an as-needed basis and are overseen by a strategic group. A review day is held annually in which members of all groups discuss progress and future

directions. This reorganization has led to more effective decision making and implementation.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- Early re-introductions of EBBs failed due to a combination of the presence of the introduced red fox, overgrazing by native and exotic herbivores, and persistent drought.
- Mt. Rothwell and Hamilton Community Parklands have remained fox free and have been successfully managed to maintain healthy EBB populations. Both are now considered to be at carrying capacity and new release sites are sought.
- The long term captive breeding program for the EBB has been intensively managed to maintain reproductive fitness and genetic diversity.
- Strong relationships between different partners in EBB conservation have been imperative for the survival of this species.

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Restoration of black-tailed prairie dogs to Vermejo Park Ranch, New Mexico, USA

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introduction

The black-tailed prairie dog (*Cynomys ludovicianus;* hereafter "prairie dog") is a fossorial, colonial, ground squirrel native to the western grasslands of the United States, southern Canada and northern Mexico. Recent estimates indicate prairie dogs occupy about 810,000 ha range-wide, representing a ~97% decline from historical occupation levels. This decline is primarily due to sylvatic plague (an exotic disease), loss of habitat and poisoning. Prairie dogs are a keystone species and numerous other grassland species, including the black-footed ferret (*Mustela nigripes*; Federally Endangered; IUCN: Endangered), burrowing owl (*Athene cunicularia*), ferruginous hawk (*Buteo regalis*), swift fox (*Vulpes velox*) and mountain plover (*Charadrius montanus*; IUCN: Near Threatened), are dependent on or are strongly associated with prairie dog colonies. Mid-19th century accounts of travelers on the Santa Fe Trail in northern New Mexico describe numerous prairie dogs on the shortgrass prairie in and around Vermejo Park Ranch (VPR). When VPR was purchased by Ted Turner in 1996, prairie dogs occupied <200 ha in a 24,280 ha shortgrass prairie landscape. Restoration of prairie dogs on VPR began in 1997 and translocation efforts began in 1999. From 1999-2006, 45 translocations were completed increasing colony acreage from 202 ha in 1997 to 3,950 ha in 2012.

Black-tailed prairie dog

Goals

- Goal 1: To restore the estimated early historic abundance of prairie dogs on VPR.
- Goal 2: To use prairie dog restoration to enhance biodiversity and improve the

status of existing imperiled species. A benchmark for the project is the establishment of a self-sustaining population (i.e., >30 breeding adults) of black-footed ferrets that meets federal recovery objectives for the species.
- Goal 3: To develop, refine and publish prairie dog translocation methods and the lessons learned during the project.
- Goal 4: To establish a large, ecologically intact and stable prairie dog complex on the shortgrass prairie that provides the opportunity for scientific research from single organism interactions to landscape level functions.

Success Indicators
- Indicator 1: A minimum of 50% of translocated prairie dogs should survive the first year post-release.
- Indicator 2: Newly established prairie dog colonies should persist and colony expansion should progress at ≥25% annually. As prairie dog colonies become established and the population increases dispersing prairie dogs should establish new colonies abrogating the need for future translocations.
- Indicator 3: Prairie dog associated species (i.e., burrowing owl, swift fox, ferruginous hawk and mountain plover) should utilize newly established colonies and populations should increase as prairie dog colonies expand.

Project Summary
Feasibility: No other North American grassland species evokes such strong emotions as does the black-tailed prairie dog. Conservationists and ecologist view prairie dogs as a native keystone species whose presence is necessary to maintain healthy grasslands with all the attendant species, assemblages and processes. Ranchers and farmers often view prairie dogs as competitors for a limited grass resource whose presence can leave the land absent palatable forage for livestock and in an early seral stage rendering it unsuitable for many agricultural purposes. In addition, the threat of listing under the Endangered Species Act has further hardened opinions. Recent efforts by several Federal agencies to compensate landowners for lands occupied by prairie dogs may help mitigate the concerns of both parties.

Prairie dogs alter the landscape upon which they live in several ways but two are most obvious. First, prairie dogs are soil engineers. They excavate deep (5 m) and extensive burrows (33 m in length), and create large mounds of soil at burrow entrances (Hoogland, 1995). Numerous mammals, birds, reptiles, amphibians and insects use prairie dog burrows as refugia. Second, prairie dogs consume and clip (non-consumptive) the vegetation around burrows. Without an unobstructed viewshed prairie dogs quickly fall prey to a host of predators. Moving east from the Rocky Mountains onto the Great Plains the climate becomes increasingly mesic and vegetation shifts from one short in stature (shortgrass prairie) to a landscape dominated by taller grasses (mixed grass prairie). Prairie dogs in the shortgrass prairie require minimal vegetative height reduction (normally via light ungulate grazing) in order to maintain a suitable viewshed, however, as grasses shift to taller representatives typical of the mixed-grass prairie, intense early season grazing by large ungulates or other treatments (i.e., burning, mowing) to reduce grass height become necessary for prairie dog colony persistence and growth.

Implementation: Standardized procedures for establishing black-tailed prairie dog colonies in unoccupied habitat (sites without pre-existing burrows) through translocation were developed and published during this project (Truett & Savage, 1998; Long *et al.*, 2006). Briefly, prairie dogs were captured in late spring through late summer using either live traps or were flushed from burrows using a water/ soap mixture. Immediately after capture, prairie dogs were transferred to an onsite indoor quarantine center and held for 1 - 2 weeks to ensure they were disease free. After the initial quarantine period, prairie dogs were moved to a prepared soft-release site and held on-site for an additional 3 - 5 days before release. Soft-releases sites were selected and prepared for occupation by prairie dogs based on the following criteria: soil type, vegetation type, proximity to project area boundaries and to other colonies, and for the potential for small colonies to expand and merge forming a single large colony. Once a site was selected 15 - 30 artificial burrows, each with a below-ground nest box buried to a depth of 1 m, were installed. Prairie dogs were then transported to the site and placed into an above-ground cage fitted over the artificial burrow, effectively preventing escape from the soft-release apparatus. Portable electric netting was installed around the site to discourage access by mesopredators (primarily badgers) and bison that often trampled above ground cages resulting in the premature release of prairie dogs. After a 3 - 5 day acclimation period, above-ground cages were removed and prairie dogs were released. Prairie dogs continued to use the artificial burrows several years after they had established natural burrows.

Post-release monitoring: Short-term post-release monitoring of translocated prairie dogs involved inspecting release sites daily until prairie dogs became accustomed to the site and began to excavate natural burrows. On most sites prairie dogs began to establish natural burrows within a day of release, however, it often took several weeks for prairie dogs to dig burrows of sufficient size, depth and complexity for them to safely occupy. At ~2 weeks post-release, we conducted visual counts to determine the number of surviving prairie dogs. For most translocations, the 2-week post release monitoring indicated a >50% retention rate. Long-term monitoring of established translocations consisted of annual areal mapping and density counts (prairie dogs/ha). Data collected from these measurements provides a reliable index to the number of prairie dogs living on VPR during a given period. Colony areal growth from 1997 - 2012 varied from 5% - 50% with an average annual increase of 22%. Prairie dog densities during this period averaged 25 prairie dogs/ha. Both areal growth and prairie dog density were strongly correlated with spring/summer precipitation. Lower than average precipitation resulted in less vegetative growth which resulted in lower prairie dog densities yet higher areal growth (prairie dog colonies expanded in search of forage). High precipitation years resulted in higher densities (higher pup production) and lower areal growth. Black-tailed prairie dog coverage on VPR has increased from 202 ha in 1997 to 3,950 ha in 2012 with a notable increase in biodiversity and abundance of associated species including black-footed ferrets.

Major difficulties faced


Post-release monitoring in typical habitat

- Establishing colonies during dry years proved to be very difficult. Excavating burrows requires substantial effort on the part of prairie dogs and during dry years the vegetation was neither sufficient nor nutritious enough to meet the energy requirements of prairie dogs. In addition, the soil tended to be "harder" in dry years further limiting the ability of prairie dogs to establish burrows.
- Badger predation during and immediately following soft-releases. Badgers would occasionally dig up below-ground soft release cages and predate the prairie dogs living in them. Badgers would also exploit the relative shallowness and simplicity of newly established burrows in the weeks immediately following release. In cases of severe predation by badgers a supplemental prairie dog release was required. Predation on recently released prairie dogs by other predators (e.g., coyotes, raptors) was more frequent but generally less damaging than that of badgers.
- Limiting prairie dog colony growth in specific areas so that colonies do not expand onto adjacent properties. Currently, neighboring landowners are supportive of our efforts to restore prairie dogs and associated species but that goodwill would undoubtedly diminish if VPR prairie dog colonies were to expand onto and colonize neighboring properties.

Major lessons learned

- Develop an open, constructive and civil relationship with all stakeholders including adjacent landowners and government agencies. To the extent possible these relationships should be developed prior to project initiation.
- Take a long-term view of the project envisioning complete success. What does the project look like in the future and what are the challenges to maintaining the program? An example from this particular project would be our rather quick and unexpected shift from managing for prairie dog colony growth to one of restricting colony growth.
- Understand and prepare for those challenges and setbacks (e.g., disease) which can reasonably be expected to occur during the different stages of the project.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- We reviewed the successes and failures of similar projects, routinely visited and communicated with other individuals and organizations involved in similar projects, and were open to new ideas. A thorough review of previous prairie dog translocation efforts, including Gunnison's (*C. Gunnisoni*) and Utah prairie dogs (*C.parvidens*), and black-tailed prairie dog habitat requirements coupled with a willingness to experiment and good record keeping allowed us to make informed decisions, detect trends and respond quickly to setbacks.
- We have fostered a good working relationship amongst all stakeholders and developed broad-based support, which is meaningful in the success of any large-scale restoration effort involving a controversial species.
- Severe prolonged drought has affected our efforts establish a self-sustaining population of black-footed ferrets on the prairie dogs at VPR. Black-footed ferret populations have fluctuated since first released in 2008 in apparent response to spring/summer precipitation levels. In 2010, >20 black-footed ferrets were identified living on VPR prairie dog colonies. Severe drought in 2011 reduced black-footed ferret populations to ~5 individuals.

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The release of northeast Bornean orangutans to Tabin Wildlife Reserve, Sabah, Malaysia

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Introduction

Bornean orangutans (*Pongo pygmaeus sp.*) are declining due to habitat destruction and fragmentation, hunting, and other human encroachment into their preferred habitats (Singleton *et al.*, 2004; Wich *et al.*, 2008), and are classified as Endangered (EN, A2c) (IUCN, 2012). A highly visible consequence of habitat loss is the presence of hundreds of displaced orangutans in rescue and rehabilitation centres throughout their range. The majority of remaining wild orangutans are located outside protected areas in forests that are exploited by humans or that are being converted for agriculture, thus it is likely that the number of orphaned animals arriving at rehabilitation centres will continue to rise. Since the early 1960s, hundreds of orangutans have passed through Sepilok Orangutan Rehabilitation Centre. Many of these individuals were subsequently released by the Sabah Wildlife Department (SWD) into Tabin Wildlife Reserve (TWR), yet nothing is known regarding re-introduction outcomes. The reserve (5°15'–5° 10'N, 118°30'–118°45'E),

Bornean orangutan © James Robins

which encompasses 1,205 km^2 of protected primary and secondary lowland dipterocarp forest, has an estimated orangutan population of 1,400 individuals, at a density of 1.26 per km^2 (Ancrenaz *et al.*, 2004). Tabin was first gazetted as a Wildlife Reserve in 1984, and is jointly managed by the Sabah Forestry and Sabah Wildlife Departments.

Goals

- Goal 1: Provide much needed data on the outcomes of re-introduced orangutans by conducting long-term regular post-release monitoring of all released individuals.
- Goal 2: Provide individual ex-captive orangutans with an opportunity for enhanced welfare through re-introduction to their natural environment.
- Goal 3: Evaluate the efficacy of current rehabilitation protocols in Sabah based on the behavioural results of rehabilitants compared to wild orangutans. In doing so, assisting rehabilitation managers in the future to produce viable release candidates.
- Goal 4: To test, and help to develop, the use of emerging technologies designed to facilitate post-release monitoring, i.e. subcutaneous telemetry transmitters.
- Goal 5: Engage local people through the delivery of an educational awareness program targeting nearby stakeholders, schools, and communities. This is designed to i) provide increased protection to the release site against illegal encroachment; ii) engender a sense of ownership and shared objectives among the local community.

Success Indicators

- Indicator 1: The collation of long-term intensive behavioral data from re-introduced orangutans in Sabah, precisely documenting re-introduction progress and outcomes.
- Indicator 2: Complete nutritional independence of rehabilitants, and the development of a healthy, stabilised post-release weight.
- Indicator 3: Demonstrably similar behavioral repertoires when compared with wild orangutans ranging in similar habitats.
- Indicator 4: Adequate integration of rehabilitants with wild orangutans to include reproduction and successful infant rearing.
- Indicator 5: The production of a larger number of viable orangutans for re-introduction through the development of improved rehabilitation protocol.
- Indicator 6: Demonstrably similar behavioral repertoires when compared with wild orangutans ranging in similar habitats.

Project Summary

Feasibility: The Tabin Orangutan Project is an orangutan post release monitoring program co-managed by Orangutan Appeal UK (OAUK) and the SWD, and was formed under the guidance of the Sabah Wildlife Advisory Panel. Field assessments conducted by Kinabatangan Orangutan Conservation Program, a local partner NGO, sought to determine the most appropriate release location within Tabin by i) identifying areas with sufficient year round food resources; ii) considering the proximity of neighbouring plantations, human settlements and roads; and iii) an area's topography and general accessibility for researchers conducting the post-release monitoring. This analysis led to the selection of an area of regenerating forest in western Tabin. The site had the highest density of fruiting trees known to be part of the orangutan's diet in Sabah, and the most diverse range of food species of five separate locations sampled. It encompasses one of the few flat areas of significant size in the area, and is dissected by a rarely used ex-logging road resulting in fast access to daily nesting locations by truck and on foot. The location is rarely ventured to by humans; the nearest settlement being the research base camp located 2.5 km away. Other sparsely populated communes close by are the SWD headquarters and a small tourist resort located 8 km away. To facilitate ongoing assessment of seasonal fluctuation of food availability, we established phenology plots where all orangutan food trees are scored by trained observers each month for their abundance of fruits, leaves, and flowers. A network of additional trails was also established to ease the tracking process.

Collecting data in the forest
© Elizabeth Winterton

Implementation: Selection of individual apes to be released was based on pre-release behavioural and medical screening. Release candidates were observed within the semi-wild confines of Sepilok/Kabili reserve during their rehabilitation phase, with orangutans deemed inadequate for release due to poor natural foraging skills, over familiarity with humans, inappropriate substrate use and locomotive patterns (e.g. too much time spent on the ground), and, hyper-sociality with conspecifics. All animals were a minimum of 6 years old at their age of release. The medical histories of all candidates were scrutinised for signs of persistent illnesses or susceptibility to disease, and they underwent periodic veterinary examinations which measured body weight, rectal temperature, pulse and breathing rate, heart and lungs auscultation, membrane colour, hydration status, and general body condition. To prevent the introduction of novel diseases into a naive ecosystem, animals were tested for potentially transferable diseases including tuberculosis, hepatitis B, and malaria. We also took blood samples for meliodosis, full blood counts and a wide biochemistry panel. Faecal smears were taken to investigate the presence of intestinal parasites, and each animal was dewormed to prevent any transfer of parasites to the release site.

School visit © James Robins

The anatomical structure of an orangutan's neck and their predominantly arboreal lifestyle preclude the use of radio collars as seen with chimpanzees (Tutin *et al.*, 2001). In attempting to overcome this most fundamental of problems, which has long constrained opportunities for thorough post-release monitoring of orangutans, the Research Institute of Wildlife Ecology in Vienna (FIWI) developed a subcutaneous radio telemetry device and implantation method for use on this project. Surgical procedures to fit these transmitters lasted approximately 25 minutes and were carried out with no adverse effects to any animal.

Post-release monitoring: Five minute nest-to-nest focal interval sampling records information on activity; social interaction; substrate use and height; and, response to human researchers. We also continuously record data on food species; plant parts eaten; feeding patch duration; and nest-building behaviour. Ranging is monitored by way of GPS track logs which provide data on each animal's home range, nest locations, and daily distance travelled. Veterinary checks of released animals replicate the periodic examinations undertaken before release. Body weight is measured wherever possible although we often experience variance in sampling timing due to the unwillingness of the animals to submit to examination. In the absence of physical symptoms, we use any significant changes in activity levels, such as apparent lethargy or reductions in normal foraging, to gauge ill-health.

Three orangutans were released in 2010 using a hard release strategy with no supplementary food offered. In 2012 experiments began with the soft release of an additional five animals whereby food is offered on an *ad-hoc* basis. Orangutans are released in small groups of 1-3 individuals. We have three confirmed outcomes so far: one animal dispersed in month six, one died in month 10, and the other died in month 12. All individuals have integrated adequately with wild orangutans, and all have experienced varying degrees of post-release weight loss in their first few months after release. One released female has given birth to an infant male and both are healthy at the time of writing. The project is ongoing.

Major difficulties faced

- Maintaining contact with exploratory and fast moving animals over steep, undulating and broken terrain.
- Limited range of radio telemetry equipment in hilly terrain and bad weather. Some transmitters also failed earlier than anticipated. The reasons for the faults may not be easily discovered as recapturing and recovering devices would be highly invasive for animals that have already been released.
- Cutting dependency on humans - even the more independent of rehabilitated orangutans may view humans as an easy source of food. We witness many instances of begging behaviour, particularly in response to increased supplementation. This is an unavoidable legacy of rehabilitated great apes spending much of their infancy reliant upon humans for most their developmental needs.
- Balancing short-term welfare with long-term chances of thriving: i) supplementing an animal's diet can be at the expense of their developing sufficient natural dietary diversity, which is all they are able to rely on once monitoring stops; ii) post-release veterinary examinations may cause undue stress and inhibit gradually developing independence - we encountered a worrying situation at one animal's routine three month examination when his pulse and temperature rose to high levels, and he became very stressed, rendering the basic parameters fundamental to a clinical assessment effectively meaningless. Equally, orangutans are incredibly stoic and may only show signs of severe illness after a condition is already well advanced, thus calling into question the efficacy of using behaviour as the primary means of assessing health.
- Inappropriate training environments to facilitate acquisition of key skills needed to survive post-release: i) twice daily food supplementation for tourism purposes in rehabilitation centres may quell the need for independent foraging and learning; ii) Tabin is a secondary regenerating forest, while the rehabilitation facility at Sepilok is located in a virgin jungle reserve. The crossover of available food species is not identical, which may explain a heavy dependence on lower quality fall back species that we have seen post-release.

Major lessons learned

- For animals that require short-term medical treatment or close observation, it is important to have a holding cage/facility located within, or very close to, the release forest. This prevents the need to transport an animal back to its original rehabilitation centre, thus limiting psychological stress and restricting the likelihood of transferring disease between two areas. While a full-time veterinary presence may not be necessary for small group releases, regular external input offers a fresh perspective on the behavioural and physical health of an animal, and is crucial to increasing survivorship. In addition, non-invasive measures of health should be pursued. Despite encountering difficulty in gaining regular access to weigh the more independent animals, a stabilised healthy weight developed during the first year after a re-introduction, combined with complete dietary independence and good health, is likely to be the most important determinant of long-term survival. Given that a reluctance to submit to physical examination should be viewed positively, it would be ideal to develop a method for non-invasive weighing in the field. Similarly, monitoring parasite loads provides another non-invasive method for assessing health. At pre-release it is important to avoid over enthusiastic pre-release worming regimes, while regulating exposure to allow some development of immunity

- Researchers should familiarise themselves with the wider release location, and try to anticipate movements away from any core areas previously identified during the pre-release phase. To maintain contact with animals, particularly in the first few months of their re-introductions, we needed to cut trails as we went. However, once more permanent trails had been established covering a larger area; we lost contact with the animals much less frequently
- Deciding when to stop following re-introduced rehabilitants is not an exact science and must be judged based on an individual's progress, and their natural desire to disperse. If animals are however not performing well, and are unable to learn from latterly re-introduced animals, they should be returned to the rehabilitation facility on welfare grounds. Given that all re-introduction mortality statistics are heavily influenced by the duration of post-release monitoring, the longer an animal can be monitored, then the truer the picture of re-introduction successes/failures and the reasons behind them
- Small group releases have enabled long-term post-release monitoring of all of our re-introduced animals so far. Depending on the number of staff available to re-introduction managers, and assuming nest-to-nest follows are conducted, we recommend that animals are followed intensively (≥three days per week). This minimises the likelihood of losing contact while also allowing for each animal's health and behavioural status to be checked on a regular basis
- Re-introduction marks the beginning of the most challenging aspect of the entire rehabilitation process. As such, post release monitoring projects involving great apes must be conducted thoroughly over several years for its data to be most valuable. To most precisely document post-release outcomes, it is vital to equip an animal with a tracking device. Today we are using radio-telemetry, although there are still limitations associated with this. Further technological development may soon produce satellite devices that last for several years, and for some rehabilitants this may dispense of the need for a potentially disruptive, and expensive, human presence on the ground

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- The project has contributed to the refinement of never before trialled implanted radio telemetry transmitters, which, in turn, has assisted researchers to stay in regular contact with all newly released animals.
- Large amounts of intensive behavioural data have for the first time been collected on the fate of individual rehabilitated orangutans.
- It is too early to assess the impact this research may have on shaping future rehabilitation protocol. More data must first be collected, analysed, and acted upon, from a larger number of orangutans, before judgement can be made on

this goal. However, the confirmed deaths of two out of three animals released during the hard release stage of the project demonstrate that in some cases rehabilitant orangutans are unable to survive without post-release support. Periodic weight loss displayed by others when not regularly supplemented also raises preliminary questions over both the suitability of the release site, and the current rehabilitation protocols in use in Sabah. In contrast, the carriage and subsequent birth of a healthy baby from a released rehabilitant mother is encouraging.
- It remains unclear how well prepared many orphaned orangutans are for thriving in a natural forest. Learning from similarly aged conspecifics or from human care givers is no substitute for an extensive mother/offspring learning period as experienced by undisturbed wild infants and juveniles.

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Release of the western subspecies of chimpanzee in Guinea, West Africa

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Introduction

Throughout their range across Africa, chimpanzees (*Pan troglodytes*) are threatened with extinction due to habitat destruction, disease and unsustainable levels of hunting and capture (IUCN 2008), in spite of being protected by national and international laws. All four known subspecies of chimpanzee (Eastern: *P. t. schweinfurthii*; Central: *P. t. troglodytes*; Nigeria-Cameroon: *P. t. ellioti*; Western: *P.t. verus*) are classified as Endangered (IUCN 2008) and listed on Appendix I of CITES. Although current total population estimates are imprecise, the second most threatened subspecies after *P. t. ellioti* is the Western subspecies (*P. t. verus*) with 21,300 - 55,600 individuals and c.50% found in Guinea (Kormos *et al.*, 2003). Unfortunately, the majority of chimpanzees in Guinea are found outside protected areas. The bushmeat and pet trade, as well as the exacerbation of human-chimpanzee conflict situations, have resulted in recent years in a significant increase in the number of orphan chimpanzees. The Chimpanzee Conservation Center (CCC), located in the north-western edge of the Mafou core area of the High Niger National Park (HNNP), is the only Pan African Sanctuary Alliance (PASA)-accredited sanctuary caring for chimpanzee orphans in Guinea. The CCC has been rehabilitating confiscated chimpanzees since 1997 and releasing selected suitable candidates since 2008.

Released chimpanzees © CCC

Goals

- Goal 1: Successfully release a group of rehabilitated chimpanzees and reinforce the numbers and genetic diversity of the wild chimpanzee population within the HNNP.

- Goal 2: Contribute to the long-term conservation of the HNNP by strengthening law enforcement activities and efforts led by government agencies and authorities locally and fostering government commitment to protecting the national park-one of two in the entire country.
- Goal 3: Increase environmental and conservation education efforts locally and nationally to influence both public-opinion and attitudes and policy-makers at the local and national level.
- Goal 4: Enhance our understanding of the release-potential of chimpanzees, the relationship between rehabilitation procedures and release success, and generally contribute to improving best practise guidelines for the rehabilitation and release or re-introduction of chimpanzees.

Success Indicators

- Indicator 1: Self-sufficient and healthy released individuals exhibiting species-specific ranging and association patterns either forming a fission-fusion social grouping of their own (eventually accommodating wild immigrant females) or having successfully integrated a wild chimpanzee community.
- Indicator 2: Successful reproduction of released individuals and infant survival rate comparable to wild conspecifics living under similar environmental and climatic conditions.
- Indicator 3: Decrease in the anthropogenic pressures and threats to the habitat and wildlife within the HNNP compared to baseline assessments pre-release.
- Indicator 4: Increase in wildlife populations within the HNNP compared to pre-release data.
- Indicator 5: Increase in environmental awareness at the local and national level contributing to the eventual demise of the pet trade and to positive changes in people's attitudes and behaviour towards chimpanzees.
- Indicator 6: Number of scientific publications, thesis, dissertations and other academic documents or media outputs based on project activities, results and findings.

Project Summary

Feasibility: Finding a suitable release site was a key step in the feasibility stage and a challenging affair since no single site in Guinea can fully comply with the IUCN Re-introduction Guidelines for Great Apes (Beck *et al.*, 2007). After careful consideration of the 1998 National Chimpanzee Survey Report by R. Ham and nationwide maps of vegetation distribution and protected areas network, four areas were selected for survey as potential release sites (Raballand, 2004). Four major selection criteria served to compare each site (Humle *et al.*, 2010). The first criterion was *habitat suitability*. The habitat had to provide i) sufficient food in quality and distribution across seasons, ii) suitable nesting sites and tree species appropriate for nesting, and iii) access to natural sources of water should water be a limiting factor. The second was *distance from human habitation and settlement*; distance to villages and settlements had to exceed 20 km, unless access was hindered by a geophysical barrier, e.g. a river. The third criterion was the *protection status of the area and current and future anthropic pressures* on

High Niger National Park survey

the local fauna, chimpanzees (if present) and the habitat. Areas where it is culturally and/or religiously taboo to kill chimpanzees and consume their meat and that already benefitted from a legal protection status were favoured over others. In areas where human activity is strictly prohibited, protection levels could be reinforced readily if necessary in collaboration with the support of national, regional and/or local government agencies. Therefore governmental support was secured early on. The fourth criterion was *the distribution and status of wild conspecifics*. Since clear risks are associated with releasing chimpanzees in an area harbouring wild conspecifics (e.g. attacks, potential resource competition, disease transmission), it was decided that the future release site was not to overlap extensively with the core area of a wild community, while being able to sustain the group of released individuals. Finally the selected site was an area in the northern part of the Mafou core area (554 km^2) in the High Niger National Park, 32 km by road from the CCC facility (Raballand, 2004). This site was distant from human settlement and presented two river networks (the Niger and the Mafou rivers) potentially restricting ranging of the released individuals into the buffer zone of the park. The environment is dominated by savanna interspersed with dry and riverine forest patches. The release site revealed a low wild chimpanzee density and peripheral usage of the release zone (30 km^2) by wild conspecifics.

Implementation: Selection of suitable release candidates was based on their long-term rehabilitation at the CCC as a social group (7 - 11 years) and individuals' ability to demonstrate species-specific social and ecological skills necessary for their survival in an environment similar to the release site. Prior to release, release candidates were screened for diseases to ensure their wellbeing upon release and to prevent disease transmission to wild conspecifics. Released candidates were also genetically screened to confirm that they belonged to the Western subspecies. A first socialized group of 6 males (1 adolescent and 5 adults) and females (1 adolescent and 5 adults) was released in June 2008 and a second group of 5 individuals (2 adults males and females with one infant-one of the males was one of the original released individuals) supplemented the first core release group in August 2011. All adults were wild-born.

For post-releasing monitoring purposes, the to-be-released chimpanzees were first equipped with mock collars 5 to 12 months prior to release (Humle *et al.*, 2010). All fully adult sized males were then equipped with VHF/GPS store-on-board/ARGOS radio collars and most of the females were fitted with simpler VHF/GPS store-on-board collars. Two adolescent chimpanzees and one adult male and female were not fitted with functional collars. A large cage and enclosure was built at the release site to facilitate release procedure. Transport was done by road in individual transport cages; released individuals were mildly to fully anaesthetized to minimise stress during transport and to cloud their sense of direction with respect to the location of the CCC facility.

Post-release monitoring: The CCC decided to implement a minimal *in situ* post-release monitoring strategy. The reasons for this were four-fold: i) promote weaning from human contact; ii) minimize potential risk of aggressive behaviour by males towards monitoring teams; iii) minimize potential risk of disease transmission from humans to chimpanzees, especially as all released individuals had been medically screened prior to release; iv) facilitate integration of released females into wild communities and promote their natural behaviour and survival skills. *In situ* monitoring thus involved i) daily location of their whereabouts either via VHF transmitters every 30 min. between 6:30 am and 7:30 pm or the remote Argos system, ii) periodic visual sightings (once every 2 - 3 months) aimed at evaluating their health condition.

Major difficulties faced

- Initial soft release protocol involving a period of acclimatization at release site in *in situ* built cage and enclosure could not be adhered to for two main reasons: i) a bushfire during months preceding the release burnt down the enclosure, and ii) not all release candidates could be moved to release cage as it was not designed to hold 12 individuals day in day out. Five males were therefore initially transported to the release cage 4 to 12 weeks prior to the release and the other seven individuals were subsequently transported to the release site the day of the release.
- Scattering of individual males and some females during the initial stages of release (within the first and second days) possibly caused by lack of complete group acclimatization at release site prior to release: this compelled retrieval missions, aimed at reuniting dispersed individuals and at returning them to the release site; during one of the missions, one adult male failed to recover from his anaesthesia due to human error. The scattering also led to losing track of three non-collared individuals. However, they were sighted a year later in a zone with wild chimpanzees; they were healthy and are presumed to be still alive.
- Ability of some released chimpanzees to cross the Niger River during the dry season: this large river was predicted to act as an impassable boundary demarcating the northern limit of the release zone. This situation inevitably raised concerns about the potential increased risk of encounter between released chimpanzees and humans in the park’s buffer zone thus compelling management to confine core release group members in the release cage for several weeks annually at the end of the dry season. The chimpanzees are then released once water levels swell back to impassable levels.

Education in schools

- Challenge in securing necessary funding for long -term post-release monitoring beyond the first year, especially linked to the expense of the sophisticated tracking collar systems used for distance monitoring. We expect post-release monitoring to continue for another three years although this will depend on future performance on release success indicators.
- Death of two new-borns among three post-release births (the first was recorded 16 months post-release): presumably by baboons widely ranging across the northern area of the Mafou core area; this group of baboons comprises more than 200 individuals; the nature of wounds on the mother (the infants' corpses were never retrieved) indicated the high probability of a baboon attack. However to date the survival rate of new-borns is 33% which is within range of wild counterparts.

Major lessons learned

- Value in i) soft group release of individuals well acquainted with one another and rehabilitated together: in spite of initial split, most released individuals now form a cohesive unit group behaving comparably to a small wild chimpanzee community and ii) releasing candidates during period of high fruit availability to maximize their initial survival and minimize food stress upon release, decreasing necessity for provisioning.
- Importance of ecological and social competence of release candidates: it is vital that release candidates are equipped with the necessary social and ecological skills to survive in release environment (familiarity with range of food items, including fallback foods during periods of fruit scarcity, locating water sources, dangers including predators such lions and leopards and potentially wild conspecifics) - two males were brought back to the CCC; these two males exhibited poorer ecological and social skills respectively compared with the other 14 candidates.
- Importance of conducting pre-release assessment of future release site and behavioural evaluations of release candidates during preparation phase. The CCC has an on-going behavioural assessment program which aims to identify suitable release candidates, to improve future assessments of rehabilitation and release success, and to help inform future release projects.
- Value of GPS store-on-board and Argos system: males ranged initially further than the females and were relocated thanks to the Argos collar system, although average transmission rate was on average only 13.2% in a relatively

open and topographically uniform environment. The downloaded GPS data contributed to our understanding of the released chimpanzees' habitat preferences, social dynamics and ranging patterns without having to observe individuals at a close distance (Humle *et al.*, 2010) - the downside to this system is the requirement to replace collars approximately every 12 months for continued post-release monitoring purposes.

- Although it is possible to release adult male chimpanzees, the release success of young adult female chimpanzees is greater than for males since young adult females are more likely to integrate wild communities (Humle *et al.*, 2010), and are less likely to incur fatal injuries from wild conspecifics should any be present (none were recorded during this project) and to take risks, e.g. in crossing challenging boundaries such as rivers.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- Self-sufficiency and adaptation of core-release group (now consisting of 8 individuals) to release zone: the core release group has settled in a defined home range within original surveyed release zone; group members demonstrate fission-fusion social dynamics and a reproductive rate comparable to wild chimpanzees.
- Released chimpanzees have adapted well to the presence of wild counterparts: Only one minor attack by wild chimpanzees on monitored release individuals was ever reported since the project began and at least one young adult female has integrated a wild chimpanzee community.
- Increased protection of the Mafou core area at least in its northern area: due to presence of monitoring staff in buffer zone and around passable river-crossing areas, in addition to increased deployment of park and local military authorities' patrols in and around core-area, and of road blocks and law enforcement initiatives, e.g. moratorium on commercial fishing along the Niger river in areas bordering the core area of the Mafou.
- Increased mobilization and awareness of the local and national authorities and local communities to the value and importance of the Niger River and the park, a site of high priority for the conservation the Western subspecies of chimpanzee (Kormos *et al.*, 2003).
- 'Insurmountable barriers' are not what they seem: annual issue with river crossing during dry season months has hampered the project's success; released chimpanzees' incursions into the buffer zone could pose a risk to humans which management is unwilling to take. The implications are severe in relation to the project's success unless the reason(s) why some of the chimpanzees (esp. males) cross the river can be identified with confidence and addressed. Bushfire management may be a possible solution, since all crossing events coincided with the presence of bushfires in release zone. Sustained education efforts specifically focused on how to behave when encountering a chimpanzee can also help alleviate these concerns; however,

these can never quite fully eliminate a risk which could jeopardise the release project.

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Preliminary observations from a welfare release of woolly monkeys in the Colombian Amazon

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Introduction

Humboldt's woolly monkey (*Lagothrix lagotricha*), the largest primate throughout most of its geographical range, is a sensitive indicator of human influence in the Upper Amazon region due to its extremely low reproductive rate and need for large areas of undisturbed primary forest. Populations were decimated in the 1960s and 1970s due to the global demand for exotic pets and spotted cat skins (the monkeys were the preferred bait in the cat traps). National laws and the CITES convention reduced the volume of exploitation, but the species is still in decline due to habitat loss and overhunting. It is categorized as VU in Colombia and VU A3cd by the IUCN.

The taxonomy of *Lagothrix* is an unresolved issue of conservation importance. The IUCN follows Groves' recognition of 4 species, based on morphological characters, while the Colombia Red List follows more recent cytological and molecular evidence consistent with a single species with four geographical subspecies.

Amacayacu National Park, like other protected areas in the Colombian Amazon, shares jurisdiction for most of its area with indigenous reserves whose inhabitants have legal rights to the traditional use of natural resources. Woollies have been locally extinguished from much of the southern part of the park.

Female Humboldt's woolly monkey
© Angélica Martínez A.

Rehabilitated individuals

© Angélica Martínez A.

Goals

- Goal 1: Establish a self-sustaining troop of woolly monkeys rescued from the wildlife trade in an area of local extinction that is now protected by the community.
- Goal 2: Consolidate and strengthen support in the local indigenous community for their ban on hunting woolly monkeys and other threatened game species in their territory.
- Goal 3: Evaluate re-introduction/ supplementation of woolly monkeys as a potential conservation tool for the management of a threatened species, for ecosystem restoration in areas of local extinction, and as an element in the campaign against illegal wildlife trafficking.
- Goal 4: Use the specific case of woolly monkeys, a threatened and ecologically important species, to facilitate the improvement of coordination and interpretation of current legal norms so that re-introduction/supplementation can be a more available and better - defined tool for species and ecosystem management in Colombia.

Success Indicators

- Indicator 1: Survival of the liberated individuals.
- Indicator 2: Species-typical behavior of the liberated individuals in terms of social interactions, foraging, use of substrate, and use of habitat.
- Indicator 3: Support, participation, and cooperation from the local indigenous community for both the maintenance of the hunting ban and for protection of the liberated troop.
- Indicator 4: Application of lessons learned in regional and national natural resource management planning.

Project Summary

Feasibility: Results reported here are from an ongoing pilot study for a possible long-term project conceived gradually as part of the evolution of the continuing discussion of natural resource use among Amacayacu National Park and the indigenous communities in its southern zone of influence. In 2004, the Mocagua Indigenous Reserve (most of which overlaps with the Park) made a collective decision to stop hunting threatened game species in its territory, with a special emphasis on the woolly monkey. The creation in the Park of a rescue center for orphaned primates confiscated from the illegal wildlife trade and a small, community-based NGO to administer this function in collaboration with the Park and the regional government agency for natural resource management (Corpoamazonía) were direct results of this agreement. At first, the rescue center simply served as an organic regional solution to the enforcement of anti-wildlife trafficking laws; activities were focused on the humane management of the confiscated victims. Healthy orphans of various primate species were free-living in natural habitat with conspecifics and with human nutritional /veterinary support.

Free-living, rehabilitated woollies begin to present special management issues as they mature - the males become dangerous and the females begin to explore widely in search of a troop to join. For this reason we decided to relocate the eight young individuals under our care to a site more isolated from human activities and gradually help them become independent. Accumulating evidence that the future diversity of Amazonian forests is highly-dependent on the seed dispersal function of robust ateline populations, that the other indigenous communities in the southern part of the park are overhunting woollies, and that the species is one of the most frequently confiscated from the illegal pet trade led us to treat this as an experiment not only in the management of confiscated individuals, but also of the wild population and a fauna-depleted ecosystem.

Implementation: In July 2010, we took an adult male and two sub-adult females to the chosen site and confined them for a few days to adjust to the change (in the small cabin built for the human support team). Then we brought up the 5 remaining individuals (a younger sub-adult female, 3 juvenile females, and a juvenile male), who were released on the spot, and freed the older ones. There was relatively little stress involved, and all the individuals stayed together, exploring and foraging as a cohesive group.

Post-release monitoring: The relocation occurred at the beginning of the season of relative scarcity of ripe fruit in the forest and as the troop began to explore we continued to provide them with food and observe them nearly continuously for six months. As the availability of fruit became greater, we began to leave them on their own for longer periods, while continuing to observe their movements and behavior regularly. During the 2011 season of fruit scarcity, when it became clear that they were losing weight we began to provide food

Juvenile woolly monkey © Angélica Martínez A.

Field staff © Angélica Martínez A.

again. During the second season of abundance, they were completely independent and no longer “central-place foragers”. In their third season of scarcity we have begun supplementing again due to an obvious deterioration in the physical condition of the male. Two individuals have disappeared and one died after we brought her back in poor health for intensive care. During the first year of this experiment there was a change in the national regulations for the management of impounded wildlife in which the release in protected areas of confiscated animals whose precise origin is unknown is prohibited, and we were no longer able to continue receiving orphans.

Woolly monkeys typically live in large, multi-male, multi-female troops whose home ranges overlap. The males are philopatric and females tend to disperse from their natal troops at around 6 years. So far there has been no reproduction in the rehabilitated group, apparently due to a “kibbutz effect”. Our original intention to create a second group of rehabilitated individuals with this in mind is no longer possible. It seems likely that the females will soon begin to search for a wild troop to join and the male will become solitary. Our conclusion from the experience is that the re-introduction of confiscated and rehabilitated woollies in areas where the natural population is locally extinct, fragmented, or significantly reduced is a viable, not harmful, and probably beneficial conservation option if long-term follow-up is possible to ease them through their first seasons of fruit scarcity. Even if the released individuals do not reproduce, their foraging restores, at least temporarily, a significant ecosystem function, i.e., seed dispersal for the many plant species with large-seeded, nondehiscent fruits dependent on these large wide-ranging primate frugivores. We recommend modification of the national norms or their interpretation so that nonarbitrary, species-specific protocols for evaluating potential risks and benefits of re-introduction can be developed and applied.

Major difficulties faced

- New national regulations for the management of impounded wildlife intended to prevent uncontrolled “dumping” of confiscated animals in effect now prevent re-introduction or supplementation as a practical option for the conservation management of protected areas in Colombia.
- There is little basic information about regional *Lagothrix* foraging ecology and our evaluation of habitat quality in the area of release, especially during the

long season of relative scarcity of ripe fleshy fruit, has been more intuitive and experiential than empirical. It is not clear whether the released individuals' difficulties in the season of fruit scarcity result from their inexperience or from the effects of selective logging for domestic use in the area, since some of the preferred timber species are also woolly monkey food plants.

Major lessons learned

- Consideration of the details of dispersal biology is critical in the long-term planning of a re-introduction. For woolly monkeys, we think a minimum of two multi-male groups is necessary, so that females reaching reproductive age can disperse from their "natal" troop.
- This project, *sensu latu*, has provided highly visible positive reinforcement for a responsible local community decision with respect to threatened game species.
- The analysis of the issues relevant to the advisability of re-introduction brought about improved understanding of the status of and increased protection for the wild population. The woolly monkey is now recognized as an "integral conservation priority" in the management plan of Amacayacu National Park as a result, and a program for monitoring the wild population has been designed and initiated. The isolation of the Colombian "trapezius" from the rest of the country has been recognized in the process; the urgent need for international action to guarantee biological connectivity within the biogeographic unit defined by the Amazon, Putumayo, and Napo rivers and the eastern cordillera of the Andes is addressed in a joint action plan of the national parks department's Amazonian subdivision and Corpoamazonía.
- The re-introduction of rehabilitated woollies appears to be a viable, not harmful, and probably beneficial possibility for conservation management, but only makes sense in the context of a comprehensive long-range strategy for species and ecosystem protection. Despite generally excellent environmental laws, Colombia lacks adequate planning and coordination mechanisms among government agencies with different functions and geographical scales of action for this to take place.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- <u>Success</u>: Total community involvement and participation from the project conception, with proactive support from national park and regional natural resource management agency.
- <u>Success:</u> Long-term commitment of those involved (community, national park, NGO), not only to reintroduction of woollies, but in general to biological conservation as a major aspect of cultural conservation, economic development, and human well-being.
- <u>Failure:</u> Top-down, arbitrary management from a national level with insufficient involvement from regional actors. In preventing the risks of pathogens,

invasive species, and exogamic depression associated with re-introduction or supplementation of wild populations in protected areas with rehabilitated individuals, the new national regulations in effect also prevent the potential benefits of increasing numbers and avoiding the loss of genetic variability associated with small and fragmented populations.

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Re-introduction of bobcats to Cumberland Island, Georgia, USA: status and lessons learned after 25 years

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Introduction

The bobcat (*Lynx rufus*) is a medium-sized spotted cat (4 - 18 kg), widely distributed in North America. Bobcats are legally harvestable in most of their range, and are currently classified as Least Concern by IUCN and listed in Appendix II of CITES, due to similarity of appearance with other spotted cat species. Bobcats in the coastal plain region of Georgia, USA, occur at densities of 0.4 - 0.6 per km^2. The most common prey of bobcats across most of their range are cottontail rabbit species (*Sylvilagus sp*). Cumberland Island is the largest of Georgia's Atlantic coastal barrier islands. Since 1972, approximately 80% of the island has been administered by the National Park Service as Cumberland Island National Seashore (CINS). The island has a subtropical climate and contains approximately 85 km^2 of upland habitat. It is accessible only by boat or small plane. Thirty-two bobcats were released on CINS during 1988 - 1989.

Bobcat on Cumberland Island © F. Whitehead

Goals

- Goal 1: Restore an extirpated native species to CINS.
- Goal 2: Reduce abundance of

herbivores (primarily white-tailed deer (*Odocoileus virginiana*), and feral hogs (*Sus scrofa*) on CINS.

- Goal 3: Increase regeneration of native vegetation (including live oaks (*Quercus virginiana*) on CINS.
- Goal 4:Test the validity of the Scent Station Index method for monitoring trends in carnivore populations.

Success Indicators

- Indicator 1: Survival of released adult bobcats on CINS.
- Indicator 2: Successful reproduction of bobcats on CINS.
- Indicator 3: Recruitment of island-born bobcats into the adult bobcat population.
- Indicator 4: Persistence of a bobcat population on Cumberland Island over time.

Project Summary

Feasibility: The 1983 Resources Management Plan for CINS identified re-introduction of extirpated species as a management objective. Bobcats are widely distributed in North America, and adapt readily to a variety of habitats and ecological conditions. Cumberland Island is located within the native range of bobcats, and they existed on the island historically until they were extirpated around 1907. Prior to the re-introduction, CINS had abundant populations of potential prey species, including white-tailed deer, feral hogs, marsh rabbits (*Sylvilagus palustris*), gray squirrels (*Sciurus carolinensis*), cotton rats (*Sigmodon hispidus*), and cotton mice (*Peromyscus gossypinus*).

An Environmental Assessment (required under the National Environmental Policy Act) was prepared prior to project implementation. Some commenters were opposed to the re-introduction of bobcats on the grounds that i) it was not desirable to control populations of herbivores or ii) it would be better to use human harvest to control herbivore populations rather than to re-establish a native carnivore. The Park Service issued a Finding of No Significant Impact, and the project was approved.

Release of a bobcat into the wild

Implementation: Adult bobcats (>1 year old) were captured using hunting dogs, leghold traps and cage traps from the coastal plain region of mainland Georgia. Captured animals were retained in a holding

Cumberland Island and Atlantic Ocean

facility on the mainland for <1 month. During this period, they were anesthetized, fitted with a very high frequency (VHF) radio-collar, and vaccinated for feline panleukopenia, rhinotracheitis, and calicivirus. Bobcats were released on CINS in groups of 4 - 6 at 1 - month intervals during October - December of 1988 and 1989. This controlled increase in population size allowed evaluation of the accuracy of the scent-station index. A total of 32 bobcats were released. All releases of bobcats were “hard releases” in which bobcats were transported to the release site and freed. One bobcat died at the holding facility when it slipped its jaw under the radio-collar. One bobcat released along the interdune meadow habitat ran into the Atlantic Ocean and swam away, and apparently drowned.

Post-release monitoring: Four graduate students and several technicians from the University of Georgia conducted three years of monitoring during and following the bobcat releases (1988 - 1991). Location and survival of all bobcats was monitored via ground and aerial radio-telemetry. Bobcats on the island were trapped using cage traps to replace radio-collars and to capture juvenile bobcats born on the island. Bobcat dens were located through intensive telemetry monitoring of females during the denning season. Food habits of bobcats were related to prey abundance by collecting bobcat scats and conducting line-transect surveys for large and medium-sized mammals and trapping webs for small mammals.

During 1997 - 1999, two graduate students from the University of Georgia conducted additional bobcat studies on Cumberland Island. A study based on a human dimensions survey of public opinion of the bobcat re-introduction found the level of knowledge about bobcats among visitors was low. Another study addressed bobcat food habits, surveys of white-tailed deer abundance, and live oak regeneration counts (Nelms, 1999). Deer harvest data from 1980 through 1997 was analyzed to compare deer condition and population structure before and following the bobcat releases.

Annual survival of the bobcats released on the island was 93% during 1988 - 1991. In the spring of 1989 at least 10 kittens were born in four bobcat litters. Three island-born bobcats were captured and radio-collared as adults in 1990. Marsh rabbits, deer, and cotton rats were major prey items during 1988 - 1991.

By 1997, marsh rabbits and deer occurred less frequently in scats relative to 1988 - 1991, and all other species occurred more frequently. Analysis of deer harvest data from 1980 - 1997 found that eviscerated body mass of deer increased after bobcats were released on CINS by an average of 5.0 - 7.6 kg for males and 2.0 - 4.9 kg for females. Deer abundance declined >50% after the re-introduction of bobcats. The number of live oak seedlings increased an average of 153.5 seedlings per 16 m^2 plot between 1990 and 1997. These changes suggest bobcats caused a trophic cascade effect through deer predation releasing oak regeneration (Diefenbach *et al.*, 2009). Visitor and deer hunter attitudes towards bobcats were basically neutral in 1997.

A project to collect and genetically analyze bobcat scats was initiated in December 2011. Nine bobcats were uniquely identified, which is likely an incomplete count of the island population. Predictions at the time of re-introduction, based on a population viability analysis, were that the population would stabilize at approximately 10 - 12 individuals (Diefenbach, 1992). Among these nine individuals, the average number of alleles observed at 12 microsatellite loci was 3.8 and the overall heterozygosity for the population was 0.519. We did not observe evidence of inbreeding and the population displayed a slight heterozygote excess (F= -0.183). Compared to genotypes obtained from six of the bobcat founders, there was a significant difference in allele frequencies (P = 0.005) between the modern and founding populations, suggesting genetic drift has occurred.

Major difficulties faced

- Environmental Assessment (EA) and Public Opinion: The initial justification of the re-introduction for the EA, to control herbivores, was a mistake (Warren *et al.*, 1990). Public support for a re-introduction for its own sake was underestimated.
- Lack of funding and agency ability to continue monitoring efforts: Intensive monitoring occurred during the first three years post-release. However, once the contract for the re-introduction and initial post-release monitoring was completed, the National Park Service did not have the resources or management priority to continue follow-up efforts. The additional research undertaken in 1997 and again in 2011 was initiated through the efforts of the principal investigators. This effect was probably compounded by staff turnover at CINS, and loss of agency knowledge about the details of the project.
- Lack of long-term storage for genetic (blood and tissue) samples: Blood samples were obtained from all of the bobcats prior to release. However, there was a fire at the facility containing blood samples at the University of Georgia, and additional samples shipped to another researcher at a different university were inadvertently destroyed.

Major lessons learned

- We believe that slow releases, whereby animals are held in captivity at the release site and allowed to leave captivity following a holding period, might have prevented the disorientation of the one bobcat that swam into the Atlantic Ocean and presumably drowned.

Captured bobcat born on island

- The use of locally adapted, wild -captured adults likely contributed to the successful survival and reproduction of the bobcats after their release. It is probably best to use experienced adults in re-introduction efforts whenever possible, and if not possible, to provide as much experience as possible to captive-bred animals prior to any release effort.
- Post-release monitoring that includes consideration of trophic-level characteristics and effects can potentially provide greater insight into project success (or failure). Despite the stated objective of controlling herbivore populations, the relatively dramatic reduction in white-tailed deer abundance was somewhat of a surprise and re-establishment of understory vegetation, including live oak seedlings, exceeded expectations.
- Had we conducted public scoping or human dimensions surveys prior to preparing the Environmental Assessment, we could have identified the diversity of public opinions that surrounded the proposed bobcat restoration. Furthermore, a proactive role with the media could have minimized misconceptions about the project and resulting controversy, and personal contacts with influential people in the local community could have allowed us to identify opposition to the project prior to formally releasing the EA. A project involving a more controversial species that potentially represents a greater threat to human safety or property (such as large carnivores) would be wise to invest considerable effort in the human dimensions aspect of the project.
- Sometime between 1999 and 2011, coyotes either were introduced or successfully immigrated onto Cumberland Island. By 2011, a year-round breeding population of coyotes existed on the island. The recent establishment of coyotes on CINS may have undesirable effects on the bobcat population or of the native prey species, and may trigger additional trophic-level effects. The establishment of the coyotes does highlight the fact that all ecosystems change over time and it may not be possible to anticipate all possible future scenarios.

Success of project

Highly Successful	Successful	Partially Successful	Failure
√			

Reason(s) for success/failure:

- The re-introduction was conducted in a protected natural area with suitable habitat, where bobcat harvest is not allowed.
- The re-introduction was conducted within the animal's native range, and with wild-caught adults.
- The re-introduction was part of management objectives for the natural area and had management support.
- Bobcats, in general, rarely conflict with perceived human interests.
- Genetic analysis of bobcat scat from the present population found moderate levels of genetic variation. This analysis suggests a shift in allele frequencies from the founding bobcats to the current population, suggesting genetic drift has occurred. Although several potentially related individuals were identified, there does not appear to be significant inbreeding occurring in the population. This bodes well for the long-term persistence of bobcats on the island.

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Re-introduction of *Manglietia longipedunculata*, an endemic and critically endangered species in China

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Introduction

Manglietia longipedunculata Q. W. Zeng & Law, a species of the genus *Manglietia* (Magnoliaceae), is only distributed in evergreen broad-leaved forest at altitudes 700 - 800 m in Mt. Nankunshan, Longmen County, Guangdong Province, China. Only one sprouting tree was found when this new species was published in 2004 (Zeng & Law, 2004). Since 2007, *M. longipedunculata*(DD) was listed in The Red List of Magnoliaceae (Cicuzza *et al.*, 2007), but there is no sufficient data about the population numbers, population size, the ecological and biological characteristics. Funded by the Botanic Gardens Conservation International (BGCI) in 2008, the field investigation and pollination biological experiments of *M. longipedunculata* during the flowering and fruiting time has been carried out by South China Botanical Garden to evaluate its conservation status and put forward the efficient conservation strategies.

Bamboo scaffolding for artificial pollination of *M. longipedunculata* © Q. W. Zeng

Goals

- Main Goal: Re-enforcing its natural population.
- Goal 1: Assessment of conservation status of *M. longipedunculata*.
- Goal 2: Assessment of threats to *M. longipedunculata*.
- Goal 3: Experimental testing of artificial pollination for *M. longipedunculata* re-introduction.
- Goal 4: Enlarging population for *M. longipedunculata*.

Success Indicators

- Indicator 1: Only one population with 11 individuals was found in approximately 100 km^2 around the neighboring area.
- Indicator 2: The main threats to *M. longipedunculata* are established.
- Indicator 3: More than 2 kg of *M. longipedunculata* seeds were collected through artificial cross-pollination.
- Indicator 4: A reinforcement population with 1,000 seedlings has been established in the original habitat of *M. longipedunculata*.
- Indicator 5: An *ex situ* population with 200 seedlings has been established in South China Botanical Garden, Guangzhou, China.

Project Summary

Feasibility: *M. longipedunculata* is only distributed in evergreen broad-leaved forests at 700 - 800 m on Mt. Nankunshan, Longmen County, Guangdong Province, China. Only one sprouting tree was found when this new species was published in 2004. This species has high ornamental value because of its beautiful tree shape, large, elegant, fragrant and white flowers. It also has high value for good timber. Though the species diversity in Mt. Nankunshan is abundant, the vegetation is mainly secondary and artificial and had been destroyed seriously before the establishment of Nankunshan Provincial Nature Reserve. The over-exploitation, natural habitats degradation and natural reproductive capability decline resulted in the threatening of this species in the wild and was listed in The Red List of Magnoliaceae as DD in 2007. Funded by BGCI since 2008, the field investigation and pollination biological experiments of *M. longipedunculata* during the flowering and fruiting time has been carried out by South China Botanical Garden to evaluate its conservation status and put forward efficient conservation strategies.

M. longipedunculata fruits

Implementation: From 2008 to 2011, comprehensive field surveys were carried out in three adjacent counties (Longmen, Conghua, and Zengcheng) and a comprehensive study about its population ecology and pollination biology was conducted, including population size and amount, pollinators, flowering characteristics, and fruit-bearing condition. So far only one extremely small population and 11 mature individuals of *M. longipedunculata* with a very narrow distribution was found in the area of about 100 km^2 in Mountain Nankunshan. In this extremely small population, inbreeding is more likely to occur and genetic variation is low, so the population is easy to be influenced by surroundings and natural disasters and finally dies out. The lack of efficient pollinators and flowering in the heavy rainy season make this species unable to develop fruits and seeds under natural pollination. So far, no seedling of *M. longipedunculata* has been found under its mature individuals until now. Its natural refreshment is very poor. Therefore, it is categorized as CR (Critically Endangered) according to the IUCN (2001) categorization (Version 3.1).

Local people, authorities and scientists at re-introduction site © Q. W. Zeng

A huge bamboo scaffolding (about 20 m high) was set up for artificial cross-pollination and pollination biological research. The endangering factors of *M. longipedunculata* are: i) The stigma receptive period of a single flower is only one day. This biological characteristic may be one of the most important factors leading to its extremely low fruit-set rate in natural conditions, ii) Lack of efficient pollinators. Beetles are major pollinators for *M. longipedunculata*, and bees never visited flowers even when they passed by them. Flowers cannot get sufficient pollens during the receptive period, which mainly caused low fruit-set rate of *M. longipedunculata* under the natural conditions (Xie *et al*., 2012). More than 2 kg of *M. longipedunculata* seeds were collected through artificial cross-pollination. About 3,000 seedlings were successfully propagated. A well managed *ex situ* collection has been established. About 1,000 seedlings were re-introduced in its native habitat in Nankunshan Nature Reserve.

Post-planting monitoring: All re-introduced plants have been managed and monitored by staff from nature reserve and they have a good growth and condition. The average height of seedlings has reached 140 cm by October, 2011, and the highest reached a height of 295 cm.

Plant growth one year after re-introduction

Major difficulties faced

- The population size is extremely small. The limited individuals cannot reproduce any individuals naturally.
- Lack of efficient visitors or flowering in the heaviest rainy season. No seeds can be collected for propagation under the natural conditions so artificial cross-pollination must be carried out for seeds.
- The tall height of *M. longipedunculata* makes artificial cross-pollination difficult. So a huge bamboo scaffolding (about 20 m high) must be set up.
- *M. longipedunculata* is a newly published species, without any existing bio-temperature information for reference. So we hired the local people to observe and record the flower and fruit bio-temperature for pollination biological research.

Major lessons learned

- Evaluating the conservation status: Prior to any restoration or re-introduction effort, the population amount, size and structure of *M. longipedunculata* and causes of the endangerment should be clear. Comprehensive field investigations and pollination biological experiments of *M. longipedunculata* during the flowering time (June) and fruiting time (September) should be conducted to evaluate its conservation status.
- Selecting the appropriate location: The site is very important for successful re-introduction. The site for re-introduction should: i) have a similar ecological environment to its natural habitat; ii) be close to the original population; iii) have the traits to be able to survive in the long-term, and iv) be convenient for management.
- Strengthening conservation awareness of local community: Local people are the custodian of plants. Their awareness of plant diversity conservation is very important for successful plant conservation. To enhance their awareness of plant diversity and environment protection, we held some stakeholder workshops in project location and distributed public outreach materials to them. People living around Nankunshan Nature Reserve were invited to attend workshops and actively communicated with government leaders and scientists. Local people are also involved in re-introduction activities and managing transplanted plants. Their awareness has been greatly enhanced through the engagement of those activities and will protect plants actively in the future.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- The active involvement of local people: The pollination biological research needs exact bio-temperature observation. Every year, local people help observe and record the flower and fruit bio-temperature of *M. longipedunculata*. With their help, we can successfully carry out the pollination biological research and collect the fruits which developed from artificial cross-pollination on time.
- Basic scientific research of target species: The re-introduction should be based on the scientific planning, appropriate technology, ecology, biology, genetics, available data or references, policy or regulations, social economics and local attitudes.
- A strong working group: Besides local communities collaborating with government, international organizations, institutes or botanical gardens is also very important for a successful re-introduction.
- Selection of seedlings for re-introduction: This is very important, including the age structure, height, quality, source and the genetic diversity of seedlings.
- Post - planting monitoring: The re-introduced plants should be managed and monitored regularly, including the monitoring of growth status, survival rate of re-introduced plants, community structure, etc.
- Stakeholder workshop and training: Students, staff in nature reserve and local people should be trained for implementing the project.

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Twenty years of Indus Delta mangroves development and rehabilitation by Sindh Forests Department, Pakistan

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Introduction

The Indus Delta mangroves represent the sixth largest mangrove block worldwide. The Delta stretches over 348 km from Karachi, Sindh, Pakistan to the India-Pakistan border. The delta is a typical fan-shaped and spread over about 617,470 ha and is characterized by 17 major creeks, innumerable minor creeks, mud flats and fringing mangroves (Qureshi, 1999). At present, 280,470 hectares mangrove forests are managed by Sindh Forest Department, 64,400 hectares by Port Qasim Authority and are declared as “Protected Forests”. Some 272,600 ha are under the control of Sindh Board of Revenue, Pakistan (Vistro, 1999). Mangroves are playing a vital role in the economy of Pakistan, besides environmental, in the shape of fisheries they harbor. Some 81,000 people living along the coastal belt use *Avicennia marina* as a major source of fuel. It is estimated that about 18,000 tons of mangrove wood is burnt annually for cooking and heating purposes. Some 6,000 camels are also herded into the mangrove forests for browsing *A. marina* leaves (Hoekstra, 1998).

The Indus Delta mangrove ecosystems is dominated by a single species; *Avicennia marina* (over 95%) followed by *Rhizophora mucronata*, *Ceriops tagal* and *Aegiceras corniculatum*. At present, Indus delta mangroves are under severe stress on account of a combination of natural and human induced factors. These factors are the drastic reduction of fresh water flow to the Delta, less addition and deposition of silt load, a tremendous increase in population along the

Avicennia marina in the Indus Delta

coastal belt resulting in the illegal cutting of mangroves for constructing residential buildings, cooking and heating. A large number of camels, cows and buffalos are also grazing within the mangroves. Seawater pollution is another major threat to mangroves. Untreated domestic sewage of Karachi city along with significant volume of untreated discharges from about 6,000 industrial units is drained in to the mangrove area. Besides, oil spills from ships, dredging of shipping channels and thermal pollution from industrial mills and thermal power plants are causing great damage to existing mangroves and also hindering the natural regeneration process.

Planting mangroves in the Indus Delta

The quality and area under mangrove forests has deteriorated and declined during the last five decades. The satellite imageries taken and surveys done at periodical intervals shows that mangrove forest have shrunk from 344,870 ha to 86,727 ha. The first survey was done by Khan in1966 shows that some 344,870 ha were under mangrove forests. Another survey conducted by Amjad and Khan (1983) estimated about 283,000 ha mangrove forests in the Indus delta. A survey done during 1983 - 1984 by Tahir Qureshi estimated about 280,470 ha under mangrove cover. After an interval of 20 years in 2003, SUPARCO prepared a mangrove vegetation map by using SPOT imageries. It was reported that area under mangrove forests have drastically shrunk to about 86,727 ha (IUCN, 2005).

Realizing an alarming situation of depletion of mangrove vegetation, Sindh Forests and Wildlife Department, Government of Sindh, Pakistan initiated nine mangrove rehabilitation/development projects from the year 1993 to 2012 with the assistance and partnership of “The World Bank, Asian Development Bank, Government of Pakistan and Government of Sindh” to mitigate the degradation process and loss of mangrove habitat. As per data compiled by the Office of Chief Conservator of Forests, Sindh, Pakistan, some 70,300 hectares have been rehabilitated/planted with local mangrove species during the last 20 years period from the year 1993 to 2012. The most fascinating aspect of these projects besides rehabilitating huge degraded areas is; setting of two new “Guinness World Records” during the year 2009 and 2013.

Goals

- Goal 1: To rehabilitate and develop mangrove habitats impacted by natural and human induced factors.
- Goal 2: To transform sparse mangrove forests into dense forests.
- Goal 3: To increase diversity of local mangrove species.
- Goal 4: To halt/minimize mangrove degradation process.
- Goal 5: To introduce social forestry in the coastal belt.
- Goal 6: To maintain plantation areas with minimum mortality.
- Goal 7: To encourage and insure participation of local communities in mangrove rehabilitation, plantation and protection activities.

Success indicators

- Indicator 1: Establish mangrove plantations on blank mudflats.
- Indicator 2: Convert sparse mangrove forests into dense forests.
- Indicator 3: Increase diversity of mangrove species.
- Indicator 4: Mortality of planted mangroves below 20%.

Project summary

Feasibility: The first mega-mangrove rehabilitation project was started during the year 1993. The project was jointly sponsored by "The World Bank and Government of Sindh". Keeping in view the success stories of this project, Sindh Forest Department launched eight more mangrove conservation and development follow on projects from the year 2000 to 2011 with the cooperation and funding of "Asian Development Bank, Government of Pakistan and Government of Sindh". Detailed field surveys of Indus delta falling in Karachi, Keti Bandar and Shah Bandar Forest Ranges were conducted by the Sindh Forest Department's staff to identify and demarcate the most suitable areas for establishing plantations.

The criteria for selecting the plantation sites were as follows:

Avicennia marina nursery

- Tidal flats with muddy substratum and natural channels where regular tidal inundations occur.
- Bare, non-vegetated areas where mangroves occurred in the past.
- Sparse natural mangrove areas.

After critical evaluation, potential sites were selected for planting and rehabilitation. The procurement of "quality planting stock" was the second most important step to execute the projects. The required planting stock was made available by three ways as follows:

- Selection and collection of healthy propagules of *R. mucronata* and *A. marina*.
- Establishment of intertidal container plants nurseries of *R. mucronata, A. marina, C. tagal* and *A. corniculatum*.
- Collecting wildings of *A. marina* from the donor sites.

Rhizophora mucronata nursery

The propagules and container plants were transported from nurseries to plantation sites by motor boats.

Implementation: This massive rehabilitation/plantation initiative was implemented through nine development projects sponsored by World Bank, Asian Development Bank, Government of Pakistan and Government of Sindh from the year 1993 to 2013. The selected areas for planting were carefully demarcated by fixing flags on the outer boundaries. Temporary holding nurseries were established near the planting sites for the storage of propagules and container plants. Before shifting of propagules and container plants from the nurseries to planting sites, each propagule and container plant was evaluated, and only healthy propagules and container plants were selected for planting. The selected propagules and container plants were put in the plastic boxes for safe handling and transportation up to temporary holding nurseries by boats. The plantation operations were carried out during low tide periods in the day time. The location of each plant was demarcated on the site. The labor and the labor supervisors were provided adequate training and knowledge on handling and planting seedlings before start of plantation operations. The plantations were established in a square shape at 3 m x 3 m spacing. Against the plantation target of 117,632 ha, 70,300 ha have been planted at various plantation sites of Karachi, Keti Bunder and Shah Bunder forest ranges from the year 1993 to 2012. Some 44,000 ha will be planted/rehabilitated within coming five years time up to year 2017.

Another milestone of these rehabilitation/plantation projects is that: two times “Guinness World Records” have been achieved during the year 2009 and 2013. On 15th July 2009, a team of 300 volunteers belonging to adjoining local communities planted 541,176 seedlings of *Rhizophora mucronata* within 24 hours time (day time) in Keti Bandar Forest Range. Once again, after a lapse of four years, a new “Guinness Record” was set on 22nd June 2013 by planting 847,275

Volunteers after setting the Guinness World Record

R. mucronata seedlings within 24 hours time (day time) by a team of 300 volunteers of local communities in Kharo Chan/Keti Bandar coastal area.

Post-planting monitoring: All the plantations established at various sites were monitored at regular basis after six months of plantation. The survival data was recorded from the permanent randomly selected plots. The survival percentage of plantations ranges from minimum 50% to as high as 90%.

Major difficulties faced

- Less and late release of funds.
- Stormy and rainy weather conditions.
- Rough high tides.
- Muddy site conditions difficult to work.
- Daily change in planting time due to change in low and high tide time.
- Limited planting season.
- Transport of saplings to planting sites during low tide period.
- Lack of adequate skilled/trained labor force.

Major lessons learned

- Site selection for mangroves plantations is most important. Survival and growth of plants depends on proper site selection.
- Predominantly bare sandy soils should not be selected for plantations.
- Plantations should not be established on high tidal mud flats.
- The survival rate is more when planting is done during the low tide period and there is no wave action.
- Involvement of local Jat community leaders is essential. Without their help and cooperation, it is very difficult task to protect young mangrove plantations.
- Local communities prefer *A. marina* plantations to *R. mucronata* plantations due to its fodder and fuel value.
- Survival and growth rate is affected by selection of planting material and site suitability of the mangrove species.
- Planting of *R. mucronata* propagules give better results as compared to container plants.
- Inbuilt mechanism of monitoring and evaluation system is the key for achieving high plantation success rate.

- Plantation maintenance funds are vital for successful establishment of mangrove plantations. Government of Sindh/Pakistan must provide adequate maintenance funds after completion of the development projects.

Success of Project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- Selection of most suitable plantation sites.
- Selection of healthy and proper sized planting stock.
- Planting operations at the correct time and planting season.
- Care in handling and transportation of plants from nursery to plantation sites.
- Effective technical guidance and supervision.
- Regular monitoring.
- Involvement and cooperation of local communities.
- Building teamwork and ownership among the labor and field staff.

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Rhizophora mucronata an extinct mangrove species re-introduced to Ras Ghanada Island, Abu Dhabi, United Arab Emirates after 100 years

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Introduction

Mangroves are one of the most important ecosystems of UAE both ecologically and economically. They support a complex aquatic food web and provide a unique habitat for a variety of bird, marine fauna and have a high aesthetic value for developing eco-tourism. Mangroves are most important spawning areas for fish and shellfish. The presence of mangroves, act as a stabilization force to protect coastline from the devastations of cyclones. *Avicennia marina* is the only native mangrove species growing in the UAE. Historical records suggest that another mangrove species, *Rhizophra mucronata*, once grew here. Due to various unknown reasons, this mangrove species became extinct about 100 years ago. *R. mucronata* is included in the “IUCN Red List of Threatened Species” as a native mangrove of UAE.

During the year 2001, Department of the President Affairs, Abu Dhabi (formerly Crown Prince, Private Management Abu Dhabi) and Environment Agency Abu Dhabi (EAD) initiated a joint comprehensive research and development program to revive back *R. mucronata*; an extinct natural heritage mangrove species of the country at Ras Ghanada Island. Keeping in view the similarity of climatic conditions, propagules of *R. mucronata* were procured from Pakistan. In the first phase, various nursery research studies on survival and growth of seedlings were conducted. In the second phase, experimental field plantations were established at Ras Ghanada Island. The plantations were established in complete natural coastal

Rhizophra mucronata plantation

environment at a Sand Hill site to evaluate the survival and growth potential of the species in the natural habitat. First monitoring and evaluation of plantations was done after five years of planting in July 2008. It was amazing to observe that out of 350 seedlings planted, 280 plants (80%) were surviving and transformed in to a beautiful plantation. The second monitoring and evaluation was done in October 2011 after an interval of three years of first monitoring and eight years after planting. It was observed that after three years, there was a 50% survival rate. However, remaining surviving plants look very healthy with dark green foliage and produced a typical prop root system. This is an indication that the surviving *R. mucronata* plants have fully adapted local site and environmental conditions and became part of ecosystems.

Rhizophra mucronata seedlings

Goals

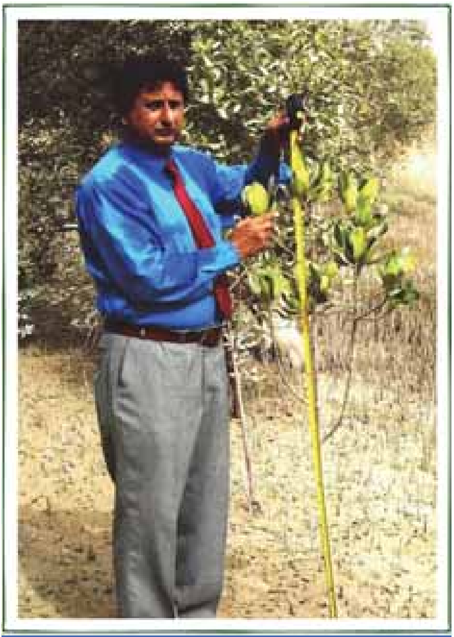
- Goal 1: Re-introduce *R. mucronata* back to Arabian Gulf waters of Abu Dhabi, UAE.
- Goal 2: Standardize appropriate nursery techniques through research and development studies for growing seedlings in the nursery.
- Goal 3: Standardize plantation techniques for establishing successful plantations.
- Goal 4: Increase biodiversity of mangrove species in the UAE.
- Goal 5: Prepare manual/guidelines for establishment of *R. mucronata* container plants nurseries and plantations in the UAE.

Success indicators

- Indicator 1: Grow healthy *R. mucronata* seedlings in the nursery.
- Indicator 2: Establish successful *R. mucronata* plantations.
- Indicator 3: Publish manual for raising mangrove container plants nurseries and mangrove plantations in the UAE.

Project summary

Feasibility: In the year 2001, “Department of the President Affairs, Abu Dhabi” (formerly Crown Prince, Private Management Abu Dhabi) and EAD started a joint venture “Mangrove Research and Development Project” to re-introduce *R. mucronata;* one of the extinct mangrove species of the Arabian Gulf



Author measuring *Rhizophora mucronata* plants

back to the coastal waters of Abu Dhabi, UAE. All the logistic, manpower and other required facilities were provided by the Department of President Affairs, Abu Dhabi. EAD provided technical expertise and arranged propagules collection and import from Pakistan.

A comprehensive research and development program focusing on development of appropriate nursery & plantation techniques was started under the supervision and guidance of the Author as follows:

- Survival and growth of seedlings in the nursery.
- Use of appropriate soil media for optimum seedling growth.
- Effect of water salinity on seedling survival, growth and physiology.
- Effect of shade on the survival and growth of seedlings.
- Use of an appropriate pot size for optimal growth of seedlings.
- Survival and growth studies on establishment of plantations.

After successful growing of *R. mucronata* seedlings in the nursery, experimental field plantations were established at selected sites during the year 2002 - 2003.

Implementation: There were three major components of the *R. mucronata* re-introduction project i) procurement of propagules from Pakistan, ii) growing sufficient number of quality seedlings in the nursery, and iii) establishment of experimental plantations. Mature *R. mucronata* propagules were procured from Dam Forest block of Baluchistan province, Pakistan with the assistance and cooperation of Sindh Forestry Department, Karachi, Pakistan and WWF, Karachi, Pakistan. A new nursery section was reserved in a screened shade-house with natural sunlight and without temperature control for conducting nursery research trials and growing container plants. The Photo-synthetically Active Radiation (PAR) inside the screened shade-house was about one-fifth of direct PAR. The fresh propagules flown from Pakistan were immediately planted in the plastic pots measuring 12.5 cm x 11.5 cm in the nursery. The propagules were watered twice a day with a mixture of seawater and freshwater in 50:50 ratio for the first two weeks. Afterwards, seedlings were watered once a day in morning time with 100% seawater.

Experimental plantations were established during the year 2003 at the Sand Hill site near the intertidal water channel with clay loam substratum. Scattered patches of young natural *A. marina* were present along the western portion of the plantation site. Eastern portion was blank without any plant growth. This was an ideal site for comparing the survival and growth behavior of *R. mucronata* on blank area and in association with *A. marina*. The selected area for planting was carefully demarcated by fixing demarcation rods on the outer boundaries. Before shifting the plants from the nursery to planting site, each plant was evaluated. Only, healthy seedlings having 50 cm height and above were selected for planting. The plantation operations were carried out during the low tide period in the day time. The location of each plant was demarcated on the site. The labor was provided adequate training on handling and planting seedlings before start of plantation operations. 350 *R. mucronata* seedlings were planted in a square shape at 3 m x 3 m spacing.

Post-planting monitoring:
Nursery: In the nursery, seedling emergence from the propagules started from the 7th day of planting and continued up to 22nd week. The seedling emergence was faster (84%) during the first 12 weeks. The seedlings attended an average height of 60 cm with mean leaf size of 42.69 cm^2 in 22 weeks.

Plantations: After 2004, no monitoring and further rehabilitation/plantation work was carried out. The first monitoring and evaluation was done in June, 2008. It was observed that out of 350 seedlings planted, 280 plants (80%) were surviving and transformed in to a beautiful plantation. Press releases of this success story were issued by the Director of the Department of the President's Affairs and were published in various Arabic and English newspapers highlighting re-introduction of *Rhizophora mucronata* after 100 years.

The second monitoring and evaluation was done in September, 2011 after three years interval. Survival and height growth data are shown in Table 1.

Table 1. Survival and Height Growth Data

Year	Surviving Plants	Average Height	Maximum Height
2008	280	1.30 m	1.70 m
2011	140	1.50 m	2.02 m

Although, during the 2008 to 2011 period, 140 (50 %) plants had died but the remaining surviving plants were very healthy. It is interesting to observe that "*R. mucronata* plants have adapted the local site conditions and are growing in the natural environment side by side with natural *A. marina*. It is interesting to observe that *A. marina,* is acting as a barrier, by protecting *R. mucronata* plants from gazelles, hot dusty winds and barnacles. The plants have attained 1.50 m - 2.02 m height. The plantations are presenting an eye catching scene of a mixed mangrove forest and a classical example of mangrove biodiversity. "These are the unique plantations of *R. mucronata* in the UAE growing only at Ras Ghanada Island".

Major difficulties faced

- Procurement of propagules from Pakistan.
- Harsh summer temperatures with dusty winds.
- Grazing pressure by gazelles.
- Presence of barnacles.

Major lessons learned

- Site selection for *R. mucronata* plantations is most critical. Survival and growth of plants depends on proper site selection.
- Predominantly bare sandy soils should not be selected for plantations.
- Plantations should not be established on low tidal mud flats.
- Healthy and appropriate sized planting stock is one of the major factors for success of mangrove rehabilitation/plantation program.
- Best planting season is November - December.
- Protection of young plants against gazelles is most important. Gazelles like to eat fleshy green leaves and also de-bark the stem by scratching with their horns and head.
- Survival rate is more when planted within the natural young stands of *A. marina*.
- *A. marina* is performing as a “motherly role” and protecting *R. mucronata* plants from gazelles, hot dusty winds and barnacles.

Success of Project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reason(s) for success/failure:

- Conducting comprehensive research to standardize nursery and plantation techniques.
- Selection of most suitable plantation sites.
- Selection of healthy and proper sized planting stock.
- Planting operations at proper time and proper planting season.
- Care in handling and transportation of plants from nursery to plantation sites.
- Planting within the natural *A. marina* young stands.
- Effective technical guidance and supervision.

Acknowledgements: The author gratefully acknowledge the Department of President Affairs, Abu Dhabi, UAE for providing facilities and assistance in establishing and maintaining *R. mucronata* nursery and plantations at Ras Ghanada Island.

First steps for the conservation translocation of the betic alder buckthorn in Eastern Spain

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Introduction

Frangula alnus subsp. *baetica* (E. Rev. & Willk.) Rivas Goday ex Devesa is a relict tree (Hampe & Arroyo, 2002) found in a few areas of Southern Spain, but having an isolated population more than 500 km far on the Eastern side of the Iberian peninsula, just sharing the borderline between the regions of Castilla-La Mancha (CLM) and Valencian Community (VC). There are some unchecked citations listing its presence in the Rif Mountains, NW Africa. It is listed as Vulnerable in the Spanish Red List of Vascular Plants (Moreno, 2008). The Eastern Spanish population is found along 20 km corridor in the deep gorges of the river Jucar. It is thought that most of the former continuous population, was severely destroyed by a big flood in 1982, which uprooted all the riverbanks destroying the ancient riverine forests. Nowadays the population are divided into two subpopulations which are 15 km from each other. These two populations are the western group (Casas de Ves, Albacete, CLM) which is estimated at over 40 specimens, and the Eastern group (Jalance, Valencia, VC), estimated at only 22

Frangula alnus baetica fruit
© E. Laguna

subrupicolous individuals, mostly placed on water-oozing crevices on tall cliffs. The plants are too far away from each other and are unable to produce fertile fruit in the wild.

Goals

- Goal 1: Fine characterization of the Valencian (VC) subpopulation, where no reproductive success is seen in the wild; searching for appropriate sites to increase the population in the future
- Goal 2: *Ex situ* production of new individuals, after setting up a plant collection. New trees are obtained after cuttings from those accessible individuals on the Valencian gorges, which are able to produce fertile ('biodiverse') seed, from cross pollination amongst different individuals.
- Goal 3: Checking the site availability for restocking the Valencian sub-population in two phases: i) by using clonal plants (to reduce the effect of genetics on the site choice); and ii) by reinforcing the chosen good sites with new plants produced from seeds (obtained *in situ*), both for the river Jucar and several close tributaries (Cabriel River).
- Goal 4: Extending the above explained strategy to the Castilla-La Mancha's (CLM) subpopulation, checking also the intermediate sites between the two sub-populations.
- Goal 5: Long-term progressive enlargement of each sub-population, and establishment of a stepped-stones connection to re-create the ancient unique population.

Success Indicators

- Indicator 1: Number of *ex-situ* produced clones and individual plants (new parent plants, produced after cuttings from the available trees in wild).
- Indicator 2: Number of new 'biodiverse' (non-clonal) seeds and plantlets obtained, including the testing of germplasm quality (germination rates, seed longevity, viability, etc.).
- Indicator 3: Number of planted individuals *in situ*, and survival rates, to be checked 2 - 4 years after plantation.
- Indicator 4: Number of established neo-populations with effective recruitment (4 - 8 years after each plantation in field).
- Indicator 5: Number of seeds produced in wild in the restored populations -

Planting alders at a wild site

testing the germplasm quality- and effective recruitment (4 - 8 years after each plantation in field).


Planted *Frangula alnus baetica*
© Pablo Ferrer

Project Summary

Feasibility: *F. alnus* subsp. *baetica* is a riverine, termophile tree (Aguilella *et al.*, 2009); close relatives and other riverine trees use to show good results for vegetative propagation (both for *in vitro* and hormoned cuttings). There is good knowledge on the ecological requirements and reproductive biology for the Southern Spanish population, in Andalusia (Hampe & Arroyo, 2002). The Valencian native plants only produce a very small amount of seeds, often sterile, consequently forcing a interim step of *ex situ* seed production, obtaining the new parent plants after cuttings or buds taken from different individuals in filed. A test experience made by the IVIA (Valencian Institute of Agrarian Research) shown good results for *in vitro* clonal propagation after buds. All the Spanish rivers are of public property, co-managed by the national and regional administration.

The sites for the new plantations in VC are unexploited wild areas placed 5 - 10 km far from the closest villages, and having difficult access for tourism related activities. The native plant sites, as well as proposed translocations sites belong to areas protected by the Natura 2000 network. Return period for the next big floods, like those causing the strong decline of the former population, is estimated at about every 500 years. The conservation of the Valencian subpopulation is made by the regional administration of the VC, having experience in germplasm conservation, seed germination and plant production with endangered plants in the CIEF (Centre for Forestry Research and Experimentation). For the Western subpopulation, the *in situ* work is done by the regional administration of CLM, having less experience and *ex situ* facilities for endangered species. A technical agreement is being established to develop the whole *ex situ* phase for both subpopulations in the CIEF nursery.

Implementation: Twenty two specimens, most of them inaccessible and scattered in 5 sub-populational groups, have been detected in the VC subpopulation, on rock crevices of the riverine tall cliffs protected from the biggest flood levels. The whole estimated extension of presence reaches about 3.6 km^2.

Picking cuttings © E. Laguna

Only a few Valencian plants produce seeds, which use to be sterile, apparently coming from self-pollination. CLM subpopulation, is still under study; the CLM trees grow on remainders of the optimal habitat, as low trees on riverbanks, forming a part of the low-tree layer of riverine forests dominated by tall willows (*Salix* spp.) and poplars (*Populus* spp.). First cuttings harvest made by climbers, and *in vitro* production using Valencian plants, both devoted to establish the *ex situ* clonal bank and to initiate the first phase of *in situ* plantations to check the sites feasibility started in 2007 - 2008. The *in vitro* production protocol was refined in 2008; and an additional successful protocol using hormoned cuttings in the nursery was also set up in 2010.

The current *ex situ* collection holds 40 individuals from five clonal lines, coming from two sub-populational groups; the genetic progressive enrichment with more lines from the remainder groups is expected after 2013. The first 'biodiverse' seed production obtained *ex situ* started in 2011, yielding more than 600 fruits and over 1,400 seeds and the production in 2012 bore over 2,500 fruits. Those first seeds show acceptable quality and they have been germinated in the laboratory in 2012. A fine germination protocol, complemented with several seed treatments and pre-treatments, is expected by 2013 - 2014. Since 2009, 320 individuals obtained from five clonal lines have been planted in the riverbanks of Jucar gorges and its close tributary Cabriel River, belonging to the Nature Park 'Hoces del Cabriel'. All the plantations can be considered as proximity reinforcements and neo-populations (see concepts in Laguna & Ferrer, 2012), close to the native sites; true reinforcements on the remainder sites on rock crevices cannot be implemented, due to their technical difficulties and the high risk of failure. Ninteen riverbank sites have been tested, the extension of presence for the whole sites reaches 766 km^2. The first enrichment of those neopopulations, from clonal plants, using plantlets produced *ex situ* from seeds has been done in the winter of 2012 - 2013. In 2012 a first cuttings harvest of some western specimens, CLM subpopulation, to initiate the parallel clonal production of new parent trees in the CIEF has also been done, and more complete campaigns are expected in the following years.

Post-planting monitoring: The plantation sites are visited every 3 - 4 months, and any incidence (phenology, predation, pests, etc.) is recorded. Eight of the 19 planted sites show good survival rates (over 80.6%); the main reasons for the unsuccessful plantations have been detected due to salinity, high fluctuation of water levels and ungulate foraging. A first blossom period of translocated plants has been observed in 2012, three years after the first translocations; however the first event of *in situ* seed production is expected in 2013.

Major difficulties faced

- Small amount of seeds, mostly unfertile and produced by self pollination, in the wild.
- Lack of knowledge of the optimal habitat requirements as former riverbanks were completely destroyed by a river flood.
- High rates of predation by Spanish ibex (*Capra hispanica*) living in the same area.
- Severe fluctuation of water level in some of the best pre-chosen sites which cannot be corrected.

Major lessons learned

- The lack of seed production is due to the long distance between wild individuals, even into each native sub-populational group. New specimens planted opposite each other show high fertility in *ex situ* conditions, and a similar behavior can be expected for the *in situ* translocated plants.
- A quick development of cultivated plants has been noticed, both after cuttings, *in vitro* explants and seeds. The new plants which are three years old can produce flowers, but the effective seed production is best made after the fourth year.
- The *ex situ* 'biodiverse' seeds (those obtained after cross pollination amongst individuals from different parents) show high rates of viability and germination.
- The translocated specimens species only grow on the first line of the local riverine vegetation (*Salix alba-Populus alba* forest, best than *Ulmus minor-Fraxinus angustifolia* community), on non-saline soils and avoiding sites showing strong fluctuations of water levels.
- The *ex situ* and *in situ* operations should follow a calendar protocol. The rooting of fresh cuttings (to initiate new lines of parent trees for *ex situ* production of new seeds) goes on best in spring, instead of autumn, using mid -diameter sizes (1.5 - 2 cm) instead of thin or thick ones. Translocations to wild habitats must be made in late autumn or early winter, instead of late winter or early spring.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reason(s) for success/failure:

- Definitive success will be considered a time the new plantlets obtained from seeds will be translocated. Provisory results are successful and allow to expect high success in a few years time.

- Failures are based on unexpected events like the recent progression of Spanish ibex population on the same sites, or unpredictable changes in water level due to dam content regulation. The translocated plants must be protected against the effect of ungulates (i.e. using metallic nets to protect them during the first years).
- The quick growth and early maturing age of the new specimens ensures the translocation success, for those sites having good conditions.
- To get native material (cuttings from rock crevices or other inaccessible sites), the *ex situ* phase is already protocoled for obtaining new seeds and plantlets in 3 - 4 years. The germination procedures are easy and quick, and the translocation sites can be well accessed.
- The species and the translocation sites are protected by Law, and the public property of the sites ensure the long-term conservation. However more fine protection measures through the designation of protected Plant Micro-Reserves (Laguna, 2001) could be required in the future.

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Conservation introduction of the Corunna daisy at Iron Knob, South Australia

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Introduction

Corunna daisy (*Brachyscome muelleri*) Sonder (Asteraceae) is a narrow-range endemic annual herb occupying a small (0.03 km^2) and extremely specialised fertile niche on steep cliff-foot slopes of the Baxter Hills (upper Eyre Peninsula) near Iron Knob in South Australia. This is a semi-arid region with a Mediterranean climate, hot summers and predominantly winter rainfall (annual average of 222 mm). The plant germinates from seed after season-breaking rains in autumn, developing a rosette of glabrous, pinnatipartite leaves. Flowers have white to pale -mauve 'petals' and develop progressively during late winter and spring on robust peduncles. Small black achenes are dispersed a short distance away from the parent plant in mid-spring, and plants senesce during late spring as temperatures rise and soil moisture declines (Jusaitis *et al*., 2003). The plant has an extremely small and localised distribution and is vulnerable to catastrophic events that could initiate rapid extinction. Its habitat is frequented by feral goats and rabbits and it is not represented in any protected reserves. The species is listed as Endangered under the Australian Commonwealth Environmental Protection and Biodiversity Conservation Act 1999 (EPBC Act), and assessed as Critically Endangered under IUCN (2001) criteria (CR B1&2ab(i)(ii)(iii)).

Goals

- Goal 1: Increase the area of occupancy of *B. muelleri* by introducing a new satellite subpopulation.
- Goal 2: Examine the natural spread of the new population from the site of introduction.

Success Indicators

- Indicator 1: Successful establishment of a new population as indicated by annual recruitment,

Brachyscome muelleri plant showing leaves, buds and flowers © M. Jusaitis flowering, reproduction and seed set over a period of 17 years following introduction.

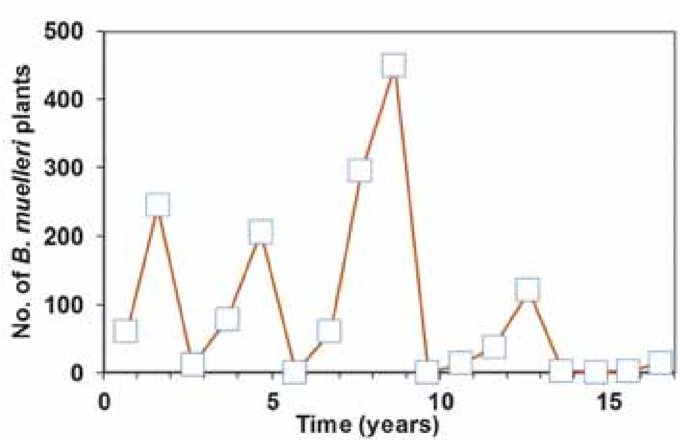
- Indicator 2: Record the maximum distance recruits spread from the original point of introduction.

Project Summary

B. muelleri seedlings were raised in the nursery during winter of 1997 and translocated to two sites in the Baxter Hills in September of that year. The first site was within the natural population, while the second (western site) was about 1.5 km NW of the natural population in an area where the plant had not been found historically, but was edaphically and floristically similar to its natural habitat. A 1 m x 1 m quadrat subdivided into 100 grids (10 cm x 10 cm) was used as a planting template and 20 *B. muelleri* seedlings (6 weeks old) were planted into a specified grid pattern within the quadrat. Three replicate plots of 20 plants were planted at each site. Recruitment was assessed annually (over 17 years), during flowering, by counting plants at all developmental stages within each quadrat. At the western site, recruits found outside the quadrats were also counted, and lateral spread of the new population was quantified by recording the maximum distance recruits had spread from the original point of translocation. Results after the first four years of monitoring were reported earlier (Jusaitis *et al*., 2004), and here I update progress up until the present, 17 years after the translocation.

Early monitoring showed abundant seedling regeneration during the first winter after translocation, followed by a decline in seedling numbers over the following two years, with numbers swelling again in the fourth year. Since then, a regular cyclical pattern of recruitment has emerged over 17 years (Figure 1). The observed cycle has a period of 3 - 4 years, and in each year immediately following a peak, seedling numbers fell dramatically. The maximum number of regenerants was observed in the eighth year after translocation, when about 450 seedlings were counted in the new subpopulation. Over the last three years plant numbers have declined, largely due to a corresponding increase in weeds, predominantly *Fumaria capreolata* and *Sisymbrium erysimoides*. This cycle did not appear to be correlated with annual rainfall, and if it is real, may be related to dormancy-cycling phenomena in the seed. Although not as marked, a similar cyclical recruitment pattern was observed concurrently in the

Figure 1. Annual recruitment of *Brachyscome muelleri* at its satellite introduction site over 16 years.



natural population. Ongoing monitoring will be required to confirm its repeatability and to understand its underlying mechanisms.

Cliff-foot slopes of the Baxter Hills, habitat of *B. muelleri* © M. Jusaitis

Seeds of *B. muelleri* are shed in the immediate vicinity of parent plants (Jusaitis *et al*., 2003). However, spread of new recruits away from the initial point of introduction was observed within the first four years of observation. Expansion of the population was predominantly in a downhill direction and seedlings were found at distances of up to 15 m down-slope from the original plots. This observation suggested that short-range gravity-assisted seed movement occurred, possibly aided by rain splash and water rivulets during periods of intense rainfall (Jusaitis *et al*., 2003). No long-distance seed dispersal mechanisms were apparent, and may explain why the species has such a restricted distribution.

Major difficulties faced

- Locating a new translocation site with similar edaphic, microclimatic and biotic features to the natural habitat of *B. muelleri* proved difficult.
- Steep cliff-foot slopes together with loose surface scree made monitoring difficult and great care was needed to minimize disturbance to the unstable soil surface layers while working at the site.
- The unique habitat requirements of *B. muelleri,* lack of long-distance dispersal mechanisms, and significant distances between suitable habitats indicate that this species is unlikely to spread substantially from its present location without human intervention.
- Population spread was limited by seed dispersal mechanisms and availability of suitable habitat.
- Herbaceous weeds and feral goats have the potential to threaten new and natural populations of *B. muelleri* (Jusaitis *et al*., 2004).

Major lessons learned

- Alternative sites within the Baxter Hills were capable of supporting an introduced *B. muelleri* population.
- Flowering, seed set, seed dispersal and natural recruitment of *B. muelleri* were all observed at the new translocation site over a period of 17 years.

- Annual recruitment was found to fluctuate in a cyclical fashion over the time of observation, with a period of roughly 3 - 4 years between peaks in seedling numbers.
- Years when no seedlings were seen did not necessarily signify that the population had become extinct.
- Seeds shed in the immediate vicinity of parent plants were able to move short distances through the action of gravity, wind and water, but long-distance dispersal was not observed.

Success of project

Highly Successful	**Successful**	Partially Successful	Failure
	√		

Reasons for success/failure:

- Successful growth and regeneration of *B. muelleri* was observed in a new satellite subpopulation over a period of 17 years.
- Years where no seedlings were seen were interspersed with years of good regeneration, suggesting that recruitment may follow a regular cyclical pattern independent of rainfall.
- Prolific quantities of seed were produced when plants were plentiful, and short-distance seed movement encouraged population spread (Jusaitis *et al*., 2003).
- The establishment of a new sub-population of *B. muelleri* has spread the risk of localized catastrophic failures and reduced the chance of genetic erosion.
- The cooperation of the owners of Corunna Station in allowing this research to be carried out on their property is gratefully acknowledged.

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Experimental translocation of the Peep Hill hop-bush into conservation reserves in the semi-arid Mallee of South Australia

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Introduction

The Peep Hill hop-bush (*Dodonaea subglandulifera* JG West) (Sapindaceae) is a dioecious, erect shrub to 2 m high. It flowers from February to June, seed capsules mature between August and December, and seeds dehisce over the warmer months of December and January (Jusaitis & Sorensen, 1994). The plant is endemic to South Australia and restricted to several disjunct populations in the semi-arid mallee areas of the South Australian Murray Darling Basin, with an outlying population near Wallaroo on the upper Yorke Peninsula (Moritz & Bickerton, 2010). It is usually found on loamy soils over limestone or slate, on private land or roadsides. None of the known wild populations occur in conservation reserves. The main threats to the species include herbivore grazing and environmental weeds. Roadside populations are usually small and isolated and can be subject to road maintenance and agricultural activities. Dodonaeas produce copious amounts of small, light pollen and are typically wind pollinated (West, 1993), so adequate cross-pollination is generally assured within sub-populations. The species is listed as Endangered under the Australian Commonwealth Environmental Protection and Biodiversity Conservation Act 1999 (EPBC Act), and assessed as Vulnerable under IUCN (2001) criteria (VU B1&2ab(iii,v)).

Dodonaea subglandulifera shoot showing leaves and fruit © B. Sorensen

Goals

- Goal 1: Safeguard the natural populations of *D. subglandulifera* by establishing new populations in two protected and secure conservation reserves.

- Goal 2: Examine the influence of founder propagule on translocation success.
- Goal 3: Examine the influence of herbivore grazing on plant establishment following direct seeding.

Success Indicators

- Indicator 1: Survival, flowering, reproduction and recruitment of *D. subglandulifera* over a period of 20 years following translocation to suitable secure sites.
- Indicator 2: The completion of an experimental translocation to evaluate the effect of founder propagule type on establishment success.
- Indicator 3: The completion of an experimental translocation to evaluate the effect of herbivores on plant establishment from direct seeding.

Project Summary

Dodonaea subglandulifera was translocated into two protected conservation parks; Yookamurra Sanctuary, secured against rabbits and feral animals by a 2 m high electric fence around its boundary, and Brookfield Conservation Park, ostensibly free-range to rabbits, goats and other vermin. These parks are respectively 8 and 15 km from the nearest wild population of *D. subglandulifera* and are the closest conserved areas with similar habitat and soil types to those in the natural range of the species. Yookamurra has a slightly higher average annual rainfall (270 mm) than Brookfield (248 mm).

Seed collected form the nearby Peep Hill population in February 1991 was used to raise seedlings for translocation and also as a source of seed for direct seeding trials. Five-month-old nursery-raised seedlings were transplanted to three sites in each park in June 1992 (30 plants per site, laid out as 3 replicates of 10 plants). At the same time, direct seeding trials were set up at a single site in each park. Seed was pre-treated by soaking in just-boiled water for 30 seconds, then kept moist until sown the next day (Sorensen & Jusaitis, 1995). Control seed was untreated and sown dry. Small plots (1 m^2) were cleared of vegetation and the soil surface was loosened using a fire rake. Seed was mixed with coarse sand and sprinkled evenly over plots (50 seed/ plot, 3 replicates). Soil was tamped down using the flat end of a fire rake. Herbivore grazing was studied by

Two rows of seedlings 3 years after transplanting at Yookamurra Sanctuary © M. Jusaitis

6 yr. old plant from the direct seeding trial at Brookfield Conservation Park © M. Jusaitis

covering a proportion of emerged seedlings with wire baskets to exclude herbivores.

Direct seeding and herbivory: Seed pretreatment with boiling water was essential for germination, and up to a maximum of 14% of pretreated seed germinated during the first spring and summer after sowing. Survival of seedlings declined over the next summer as a result of moisture stress, leaving only 15% of emerged seedlings surviving at Yookamurra and 30% at Brookfield. Seedlings covered with wire baskets grew significantly taller at both sites than those exposed to grazers. At Brookfield, unprotected seedlings were largely destroyed as a result of grazing within 2 - 3 years, while at Yookamurra survival was not affected by grazing because of ongoing vermin control and effective exclusion fencing surrounding the sanctuary. Survival remained constant at these levels for the next 11 years, until a series of severe drought years (2006 - 2009) resulted in the loss of all remaining seedlings at both sites by September 2008. Although seedlings had reached heights of 600 mm by this time, they had not developed sufficiently robust root systems to withstand several consecutive years of below average rainfall. Fruit was observed on Brookfield plants 10 years after sowing.

Transplants: Seedling transplants of *D. subglandulifera* showed an average survival of 70% in Yookamurra Sanctuary, and between 50% - 60% at Brookfield after 10 yrs. Growth differences between sites at Yookamurra were attributed to edaphic factors, and the reduced survival at Brookfield was largely a result of herbivore grazing and burrowing activities evident in the park. At one translocation site at Brookfield no plants survived beyond their second year due to herbivore activity. Transplants at both parks first developed fruit in their fourth year. By 14 years (2006) after transplanting, plants at Yookamurra and Brookfield had reached average heights of 600 mm and 400 mm respectively. When the drought came in 2006, the Yookamurra site lost nearly 70% of its surviving plants, while Brookfield lost only 15% as a result of severe moisture stress. This interesting result may be due to the larger Yookamurra plants requiring more moisture to support their larger leaf canopies than the smaller Brookfield plants, thus enabling the latter to survive a longer period of soil water deficit. Dieback generally occurred from the shoot tips, down, and in a few instances apparently dead plants were found to resprout from basal stems and to recover once good rains returned.

In summary, transplants formed the more effective founder propagule for this species. While grazing was not a significant threat to larger plants, it did affect small seedlings developing through their early growth stages, particularly at Brookfield where herbivores were more prevalent. All seedlings developing as a result of the direct seeding trial were lost during the drought of 2006 - 2009. Of the original transplanted seedlings, 13% still survive at Yookamurra and 45% survive at Brookfield 20 years after translocation. While Yookamurra encouraged more rapid early growth of transplants possibly due to its higher rainfall, these plants also suffered more as a result of the prolonged drought than did the Brookfield planting. Although plants were observed to flower and set fruit, no new recruits were seen at either site over the course of 20 years.

Major difficulties faced

- Lack of suitable conserved habitat within the population range of *D. subglandulifera* in South Australia resulted in translocation occurring outside it.
- Both Yookamurra and Brookfield parks are in lower rainfall areas than the naturally occurring populations of *D. subglandulifera*, and this has contributed to lower growth and survival rates and higher plant losses than expected in natural populations.
- Several consecutive years of below average rainfall (2006 - 2009) contributed to severe plant losses at both translocation sites.
- Plant losses were experienced at Brookfield due to herbivore grazing and burrowing activities.

Major lessons learned

- Transplanted *D. subglandulifera* seedlings were able to establish and survive more successfully than directly sown seed, as evidenced by the complete extermination of direct sown seedlings as a result of the 2006 - 2009 drought.
- Protection of small seedlings from the effects of herbivores improved establishment success, particularly at Brookfield where rabbits, goats and kangaroos were more prevalent.
- Larger, established plants were not susceptible to grazing, although dieback of shoot tips was observed in response to severe water stress.
- Lower growth rates, delayed flowering, reduced seed set, and plant losses due to drought and water stress suggest that the translocation sites chosen in these two parks may be less than optimal habitats for *D. subglandulifera*.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reasons for success/failure:

- Good survival, growth, flowering and fruiting of transplants was observed at three sites in Yookamurra Sanctuary and at two sites in Brookfield Conservation park after 20 years, although no new recruitment was observed over that time.

- The highly effective 2 m high electric fence around the Yookamurra Sanctuary was crucial to exclude feral animals from grazing or disrupting transplants.
- Demonstrated that the source of founder propagule (seed *vs.* transplants) had a significant influence on translocation success.
- Severe, prolonged drought between 2006 and 2009 caused total loss of emerged plants in direct-seeded trials, and significant losses in transplant trials. Losses may have been exacerbated by the already low average annual rainfall at these two sites.
- Preliminary research on propagation methods for *D. subglandulifera* (Sorensen & Jusaitis, 1995) enabled large numbers of plants to be propagated when required for translocation.
- Commitment to long-term management and monitoring of translocated populations ensured goals were successfully achieved.
- The commitment of the Australian Wildlife Conservancy and Conservation Volunteers Australia to maintaining Yookamurra Sanctuary and Brookfield Conservation Park respectively as conservation sanctuaries for wildlife, and for supporting research and education on threatened species is acknowledged.

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Conservation translocation of the large-headed daisy to Mount Bold, South Australia

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Introduction

Brachyscome diversifolia (Graham ex Hook.) Fischer and Meyer (Compositae) is a perennial daisy with tufted leaves and long peduncles up to 30 cm, terminating in white flowers. Flowering occurs between spring and early summer and flowers readily set seed (Salkin *et al*., 1995). *Brachyscome diversifolia* occurs in open woodland forest on steep rocky slopes and gullies, and on coastal cliffs (Salkin *et al*., 1995). There are two small extant populations of *B. diversifolia* in South Australia, one at Scott Creek Conservation Park and the other at Ironbank, both in the Adelaide Hills within 20 km of each other. Populations are restricted in South Australia because of the species' specialised habitat preferences. Potential threats to the species include grazing by slugs and snails (Salkin *et al*., 1995), weed invasion (Wilson & Bignall, 2009), trampling of populations along hiking trails, and limited suitable habitat. The species is endangered in South Australia (National Parks and Wildlife Act 1972). It also occurs in New South Wales, Victoria, and Tasmania, but is not listed as threatened in these states. The South Australian population has biogeographic and genetic significance as it represents the western-

Translocants flowering and setting seed 6 months after transplanting © M. Jusaitis most outlier for the species, significantly disjunct from its nearest neighbour in Victoria.

Grazing damage to *Brachyscome diversifolia* leaves three months after planting © M. Jusaitis

Goals

- Goal 1: Extend the natural population of *B. diversifolia* by re-introducing it to a putative historic site.
- Goal 2: Determine any threats to the survival and establishment of the species.
- Goal 3: Examine the influence of Ambiol seed-pretreatment on translocation success.

Success Indicators

- Indicator 1: Survival, reproduction and recruitment of *B. diversifolia* over ten years following translocation.
- Indicator 2: The completion of monitoring to ascertain any threats to survival and growth of translocants.
- Indicator 3: Increased knowledge of how Ambiol might be used to improve the survival and health of translocants.

Project Summary

Feasibility: The translocation site was located in the water catchment of Mount Bold Reservoir, a secure natural bush site protected from direct human disturbance. The site was chosen because of similarities in slope, aspect and vegetation to remnant habitat sites, and because the species had been previously recorded at the Mount Bold water catchment (State Herbarium of South Australia records, 1993 & 1994). The translocation site has a steep, rocky south-facing slope supporting an open woodland forest above a thick understorey of grasses and herbs. The extreme steepness of the cliff made the process of planting and monitoring quite difficult.

Implementation: Translocants were raised in a glasshouse from seed collected from a naturally occurring population at Scott Creek Conservation Park, approximately 10 km from the translocation site. The existing population at Scott Creek is very small (<200 plants), and one of only two naturally occurring populations in South Australia.

As part of the translocation, Ambiol (2-methyl-4-[diethylaminomethyl]-5-hydroxybenzimidazol dihydrochloride) was tested to determine if it could improve survival and growth of translocants. Ambiol has been effective in improving

Translocation site in the Mt. Bold Reservoir water catchment area © R. Aleman

seedling growth and relieving drought stress of agricultural plants such as tomato (MacDonald *et al*., 2010) and carrot (Lada *et al*., 2005), and North American pine trees (Borsos-Matovina and Blake, 2001), but has not been tested on Australian native plants. To test the effect of Ambiol, seeds were pretreated by soaking in 10 mg/L Ambiol for 24 hours before sowing. An equivalent number of seeds were presoaked for the same length of time in water. Control seeds received neither pretreatment. The seeds were then germinated in petri dishes for eight weeks before being transplanted to 50 mm (diameter) tubes containing commercial potting mix and grown on in a glasshouse until seedlings were vigorous enough (10 weeks old) to survive at the translocation site.

Planting at the translocation site occurred in June 2011, during the wet season when soil was moist. Seedlings (10 weeks old) were planted in a prescribed planting pattern using a 1 m^2 quadrat subdivided into 100 grids (10 cm x 10 cm). Five plants of each pretreatment (Ambiol, water, no pretreatment control) were randomized within each quadrat, and quadrats were replicated six times over an area of approximately 25 m x 10 m. At planting, and thereafter at approximately three-monthly intervals, rosette diameter, plant height (to highest emerged peduncle if plants were in bud), number of healthy leaves, number of peduncles, grazing damage, and plant health rating (scale of 1-5, 1 = dead, 5 = alive and completely healthy) were recorded.

Post-translocation monitoring: A year after translocation, 81% of all translocants survived. Most of the plants experienced their usual die-back in summer as the soil dried out, and re-sprouted again from root stocks following opening rains in autumn. The plants derived from seed pretreated with Ambiol failed to show any significant difference in survival, growth or health compared to water-pretreated or control plants, all three groups performing equally well. Five months after planting (November), most plants had produced flowers and were setting seeds. However, no new recruitment was observed during the following winter.

Although about 60% of plants were observed with some minor grazing damage, this did not seem to affect plant survival. The damage was possibly caused by snails, slugs or caterpillars, but plants seemed to be able to withstand a small degree of leaf damage by regrowing new leaves from the central meristem. Despite the observed grazing, 81% of plants survived after 56 weeks, suggesting that grazing does not constitute an immediate threat to survival. So far the site appears to be an ideal weed-free habitat for the species. We intend to continue monitoring the trial and time will tell whether the long-term sustainability of the translocation will be assured through recruitment of new individuals.

Major difficulties faced

- The extreme steepness of the slope at the translocation site made planting and monitoring difficult.
- Invertebrate predation was observed on some plants over a year of monitoring, and will need to be monitored to ensure it does not become a long-term problem.
- Recruitment was difficult to assess in the early stages because of a dense herbaceous layer. New recruits may only become evident after they reach a certain size and emerge from this layer.

Major lessons learned

- Selecting a translocation site as similar as possible (aspect, slope, vegetation associations) to the original habitat was very important for the success of this translocation.
- Invertebrate grazing did not significantly affect survival of plants over one year, but may cause long-term issues and will continue to be monitored.
- Recruitment of new plants may be a slow process, as no new individuals were seen one year after planting, even though flowering and seed production were prolific during the previous season.
- Ambiol applied as a seed pretreatment had no effect on survival, growth or health of plants. This may be because the translocation site received ample rainfall and plants were not subjected to water stress. Preconditioning seed with Ambiol is thought to promote drought tolerance in many plants.

Success of project

Highly Successful	Successful	**Partially Successful**	Failure
		√	

Reasons for success/failure:

- The survival rate after the first year was good (81%), and 97% of all surviving plants were healthy, even though some invertebrate grazing was observed. It may be too early to observe recruitment yet, as young plants may remain hidden beneath the herbaceous layer until they emerge. Long-term success is predicated on successful recruitment and this will continue to be monitored.
- Selection of a suitable habitat for the translocation site was critical to success.
- Use of locally-sourced seed may have contributed to successful plant establishment because of similar climate and habitat features to the seedlings' original habitat.

- The commitment of the South Australian Water Corporation to maintaining the Mount Bold catchment as a conservation zone and for supporting research on threatened species is acknowledged.

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Conservation introduction of Bakersfield cactus in the southern San Joaquin Valley, California, USA

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Introduction

Bakersfield cactus (*Opuntia basilaris var. treleasei*) is a perennial stem succulent in the cactus family (Cactaceae) and is endemic to Kern County in central California, USA. Bakersfield cactus occurs on floodplains, low rolling hills, ridges, and bluffs (USFWS, 1998). Soils commonly are well-drained and are sandy or gravelly with little silt, clay, or organic matter, and may contain cobbles or boulders. Plant communities include chenopod scrub, grasslands, and dry oak woodlands (USFWS, 1998 & 2011). Many sites with Bakersfield cactus have been converted to agricultural and urban uses and petroleum production. Approximately one-third of known populations have been extirpated; the remaining populations are fragmented and generally occur on small parcels (Cypher *et al.*, 2011b). Populations continue to be lost, and habitat conditions are being degraded for some remaining populations. Consequently, the taxon is listed as Federal and California Endangered. This taxon currently has no IUCN status.

The establishment of additional populations could contribute significantly to the conservation and ultimate recovery of Bakersfield cactus. Translocation and introduction is a potential strategy for establishing new populations for this species. We attempted two experimental translocations using shed pads and small plants in an effort to establish new populations in suitable but unoccupied habitat.

Cactus in native habitat showing threats from non-native grass and agricultural conversion


Co-authors translocating a Bakersfield cactus plant

Goals

- Goal 1: Establish new populations of Bakersfield cactus in suitable but unoccupied habitat.
- Goal 2: Evaluate methods for effectively translocating and establishing Bakersfield cactus.

Success Indicators

- Indicator 1: Rooting and survival of translocated cactus pads for at least 2 years.
- Indicator 2: Survival of translocated plants for at least 2 years.
- Indicator 3: Growth and survival of new pads on translocated pads and plants.
- Indicator 4: Development of guidelines and procedures for future translocation efforts.

Project Summary

Feasibility: Feasibility issues for this project included biological, environmental and socio-political considerations. Biological feasibility was deemed to be high. Translocation, particularly of pads (i.e., stem segments, cladodes), was considered to have a high probability of success because Bakersfield cactus, typical of many cacti, easily reproduces vegetatively through the shedding and rooting of pads. Furthermore, pads and plants were going to be planted in optimal microsites in that the site would be cleared of competitors (Cypher & Fiehler, 2006), the soil bed would be loosened to facilitate moisture penetration and root development, and solid ground contact would be ensured to further facilitate rooting. Finally, Bakersfield cactus had been translocated successfully previously, although in small numbers.

Environmental feasibility was also deemed to be high because the two introduction sites chosen were located within the historic range of Bakersfield cactus, and indeed, were only approximately 500 m and 100 m, respectively, from existing cactus populations. Topography and soils at both sites were consistent with conditions occurring within existing populations. Bakersfield cactus is absent from numerous locations with apparently suitable conditions, probably due to past disturbances (now mitigated) or dispersal limitations (e.g. movement uphill is probably rare). Thus, the potential for the remaining habitat to support more populations is high.

Socio-political considerations may present the greatest challenge to feasibility. In general, there is considerable resistance in the region to expanding the distribution of Bakersfield cactus, or any other rare species, due to regulatory restrictions concomitant with the presence of such species. Thus, the potential for establishing new populations is limited to lands and landowners dedicated to the protection of natural resources. Also, the agencies overseeing endangered species conservation further restrict any translocations to lands that will be legally conserved in perpetuity. Despite these restrictions, a number of sites in the region meet all the criteria above (e.g., suitable habitat, permanent protection, willing landowners) and potentially are available for the establishment of cactus populations.

Implementation: Bakersfield cactus pads and plants were collected from a population approximately 4.5 km and 2.5 km, respectively, from the two introduction sites. For the first effort conducted in October 2009 (Cypher *et al.*, 2011a), 10 small plants and 25 shed pads were collected from a protected portion of the source population. For the second effort conducted in January 2011, pads and partial plants were collected from an unprotected portion of the source population that was undergoing active conversion to citrus orchards. At the introduction sites, all vegetation was cleared from a 0.25 m^2 area for each pad or plant. The soil at the site was loosened to a depth of ca. 20 cm - 30 cm. In both efforts, plants were installed by hand-digging a small excavation and then installing the plant and filling the hole with local soil. In the first effort, pads were laid horizontally (flat side down) at each planting site and secured with a wooden skewer. In the second effort, the pads were partially buried in the soil in one of three orientations: horizontal, vertical (upright), or on edge. All pads and plants were then thoroughly watered

Issues considered included precipitation, competition, and cattle. The region is arid and precipitation is unpredictable. On average, the region receives ca. 16 cm of precipitation annually. Therefore, we occasionally provided supplemental water to plants at each site as we deemed necessary. The first site was watered two times before soil moisture from natural precipitation was deemed sufficient. The second site was watered two times, including one during the summer following planting. In addition to clearing vegetation prior to planting, we also hand-pulled vegetation from around cactus plants during the first year to reduce competition for soil moisture. To prevent injury

Cactus under cattle guard - note flower bud, new pads and non-native grass competitors to plants from cattle, we inserted two bent steel bars over plants to discourage trampling by cows.

Post-planting monitoring: At both introduction sites, success was monitored by visiting each site periodically and determining whether plants were still alive, and whether any flowering or vegetative reproduction had occurred. At the first site, nine of the 10 translocated plants were still alive (90% survival) after 34 months. However, only four of the 25 translocated pads were still alive (16% survival). The four pads that survived were among the heavier pads translocated. Some plants had flowered, and at least two plants had shed pads that subsequently rooted. Some cattle damage was noted, but only on plants that had not been protected with the steel bars. At the second introduction site, 100% of translocated plants were still alive after 25 months and several had flowered. Among partially buried pads, those with a vertical orientation had higher survival rates (mean 89%), than those laid horizontally (63%) or on edge (60%).

Major difficulties faced

- Agency restrictions on size and quantity of cactus removed from the source population.
- Plants with many pads disarticulated during transport or planting.
- Survival of transplants, especially pads, during the dry summer season.
- Invasive plants colonizing the cleared bedding areas.

Major lessons learned

- Small plants are preferable to pads for translocation. Plants that are too large disarticulate during the process, and pads have lower survival rates.
- Pads should be placed upright in pots and allowed to develop roots prior to outplanting. This has been successfully implemented by a local preserve that learned from our effort, and five new populations of Bakersfield cactus have been established at the preserve.
- Larger, heavier pads tend to grow faster and have higher survival rates.
- Control of competitors is essential, both before and after transplanting.
- Watering during the first one or two summers increases survival rates.

Success of project

Highly Successful	Successful	Partially Successful	Failure
√			

Reason(s) for success/failure:

- Anticipating and addressing many of the potential difficulties prior to implementation.
- Relative ease of propagating succulents compared to herbaceous or woody plants.

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NOTES

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