RE-INTRODUCTION

NEWS

Newsletter of the Re-introduction Specialist Group of IUCN's Species Survival Commission (SSC)

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Edited by
Pritpal S. Soorae & Philip J. Seddon

RE-INTRODUCTION *NEWS* No. 19 November 2000

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COVER PHOTO:
An adult takahe *Porphyrio hochstetteri* (formerly *mantelli*), a large flightless gallinule endemic to New Zealand, which have been successfully introduced from its native alpine habitat to several 'predator-free' lowland islands.
© Angus Small

Letter from the Chairman, IUCN/SSC Re-introduction Specialist Group, Environmental Research & Wildlife Development Agency, UAE
DR. FREDERIC J. LAUNAY

As new in-coming Chairman of the IUCN/SSC Reintroduction Specialist Group, it is with great pleasure that my first letter is for a special bird issue.

Being a "bird person" myself, and having been involved in the early stages of reintroduction projects in Saudi Arabia, it is exciting that this newsletter is sponsored by the National Commission for Wildlife Conservation and Development (NCWCD), Saudi Arabia for which I was previously working!

This bumper issue at 56 pages long reflects the numbers of quality bird reintroduction projects worldwide, but it also highlights the necessity to have a very critical eye on each project, its goals, methods and success or failures. The range of species, habitats and context in which re-introductions or translocations are done is huge and it is nearly impossible to provide the miracle recipe for a successful reintroduction.

Several contributors highlight the constant revision of methods used to release birds back to the wild. The assessment of the success of a reintroduction project is also something that needs to be constantly revisited, as shown (sadly) recently by the example of the Arabian Oryx in Oman. A success story can very quickly become a disaster and biologists, conservationists and decision-makers should always be very cautious in monitoring the success of any reintroduction projects. Gathering as many examples of first-hand reports and experiences is a way to draw attention on the successes, limitations and failures of others; and can be used to extract basic rules that apply across the board and the long-term commitments from all involved parties in reintroduction projects.

I would like to conclude by thanking all contributors to this special issue and for sharing their experiences. I would also like to give a very special thank to Phil Seddon for his dedication and commitment in advocating sound bird reintroduction practice and of course to Pritpal Soorae "Micky" who successfully completed the publication of this issue, in between an Africa to Asia migration.

Letter from the Secretary General, National Commission for Wildlife Conservation & Development, Saudi Arabia
PROF. DR. ABDULAZIZ H. ABUZINADA

Throughout the world birds are used as highly visible, charismatic focal points for conservation efforts. Key species provide accessible indicators of ecosystem health, while avifaunal communities have been used to identify regions of high general biological diversity. Despite the tremendous global interest, at levels ranging from novice bird-watchers through to committed conservation scientists, birds are not exempt from the current extinction crisis.

Whereas some species can be sustained in the wild through dedicated programmes of habitat protection and public awareness, populations of others are threatened to such an extent that more drastic conservation measures are required. The conservation tool of re-introduction, whether involving the release of captive-bred birds or translocation from the wild, aims to restore free-living populations in suitable areas of habitat. This issue of Re-introduction News compiles accounts of a selection, but by no means all, of the on-going bird re-introduction projects. The aim was to provide a snap-shot of current techniques used in the re-establishment of bird populations. This is not simply a 'good news" issue of success stories and smooth progress, although there is much here to be cause for optimism. Contributors were encouraged to review critically the successes and failures of their programmes - assessing also why particular approaches worked or did not. It is only with critical and open analyses such as these that we can hope to move forward in the development of more effective and more taxon-specific methods to halt and address avian population declines.

Saudi Arabia's National Commission for Wildlife Conservation and Development enjoys a fruitful association with the Re-introduction Specialist Group. It is therefore with great pleasure that we were able to sponsor this special issue, and we hope that you may find something of interest and use presented here.

A. H. Abuzinada

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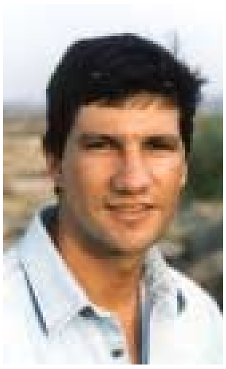
INTRODUCTION BY RSG BIRD SECTION CHAIR

Philip Seddon



Recourse to the re-introduction of any taxa often implies that numbers in the wild are at critically low levels. The challenge therefore facing any re-introduction practitioner is to balance the urgent need to undertake effective management action, against the equally pressing need to develop the necessary techniques. On top of this, any given release will most likely involve only low numbers of animals, and release techniques often must be developed from scratch. Nevertheless, it is only through carefully designed trials and experiments, along with adequate post-release monitoring that we can hope to learn as we proceed. In this way we can work towards more taxon-specific re-introduction guidelines so that as the list of re-introduction candidate species inevitably grows, fewer and fewer projects will have to start from first principles.

It was with such guidelines in mind that this Bird Issue of Re-introduction News was compiled. My co-editor, Pritpal S. Soorae (Micky) and I sought to provide a selection of bird re-introduction projects that had been critically reviewed by the field and management staff involved in each. The response to our initial requests for submissions was almost overwhelming, but the standard of the reviews provided was so high as to justify the production of a “bumper issue”. Large as it may be however, this newsletter contains only a sample of current projects. We aimed to achieve as wide a taxonomic and geographic coverage as possible, and have managed to compile 24 project reports, dealing with 45 species, from 19 families and 10 orders. Eleven countries are represented, with only a slight bias towards the southern hemisphere, due to the relatively large number of Australasian projects.

Space does not allow a detailed synthesis of the projects, but some general points are worth making. In the light of earlier comments about experimental approaches, it is gratifying to see that attempts have been made, often in the face of minimal sample sizes, to design each release so as to ensure that concrete conclusions could be drawn. This type of systematic approach is exemplified by the use of less threatened substitute species, such as the releases of Hispaniolan parrots as a model for Puerto-Rican parrots. Nothing will be learnt without adequate follow-up after releases, so it was pleasing to find that all the projects presented here undertook detailed post-release monitoring, using a variety of methods including radio-tags, banding, bells, and call-back, according to constraints of terrain and budget. Soft release techniques were not assumed to be the most appropriate in all situations; hard releases yielded better results in a number of projects, including those for hihi, kaki and malleefowl.

Active predator control or the use of predator-free islands were a feature of a number of projects, notably those for pink pigeons in Mauritius, Hawaiian passerines, and teal and shore plover in New Zealand. The provision of supplementary food was seen to be important for several species, ranging from the Aldabra white-throated rail, to great bustards in Germany. In the case of more intelligent species, such as the echo parakeet, it was possible to train birds to use special feed hoppers and to associate a whistle with food.

Veterinary screening of both release candidates and wild populations was a feature of many programs, such as that for wattled cranes in South Africa. Detailed post mortem examination of kaki actually identified a dietary deficiency that may have increased susceptibility to predation.

Micky and I would like to congratulate all the contributors and their colleagues for the excellent work they are doing, and to thank them for providing such timely and worthwhile reviews. We hope that this issue will further stimulate communication between members of the world wide bird re-introduction community.

We would also like to thank the following people for their help during the various stages of newsletter production: Hany Tatwany, Mark Lawrence, Amani Issa and Yolanda van Heezik.

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AFRICA

Re-Introduction of the Aldabra white-throated rail, Seychelles

Ron Gerlach & Ross Wanless

Introduction

Aldabra Atoll (Lat. 9° 20', Long. 46° 12') is a large coral atoll, consisting of four major islands; Picard, Grande Terre, Polymnie and Malabar rimming an extensive, tidal lagoon and fringed by an intact reef. Thanks to interventions by, among others, Charles Darwin and the Royal Society, Aldabra has remained relatively pristine and ecologically intact, unlike most islands in the Indian Ocean. Aldabra has not had a single alien bird introduced, and no avian extinctions have been directly attributable to human influences, which is totally unique for a system as big as Aldabra. In 1982, it was declared a UNESCO World Heritage Site, and the conservation of its ecosystems and biodiversity has become an important responsibility for the Seychelles Islands Foundation (SIF) which is the government organization in charge of Aldabra.
The Indian Ocean islands were once renowned for flightless birds; Aldabra is home to the last: the Aldabra White-throated Rail *Dryolimnas cuvieri aldabranus*. The limited distribution of this species has been a source of concern and its conservation remains an urgent priority. Despite the dubious distinction of being the last survivor of the flightless birds in western Indian Ocean region, the rail has shown remarkable resilience. The only genuine threat at present is the presence of feral domestic cats *Felis catus* on Aldabra. The disjunct distribution between cats and rails, and historical records of rails on Picard Island before cats colonized it, are strongly suggestive that cats are responsible for the absence of rails from Grande Terre, the largest island of the atoll. The continued presence of cats on Grande Terre (cats have been eradicated from Picard) is sobering, and is part of the motivation behind calls for re-introduction and captive breeding of rails.

Rails have been recorded in all habitats on Aldabra, but impenetrable, *Pemphis acidula*–dominated scrub supports the greatest density of rails. Being flightless, rails feed exclusively on the ground, occupying the niche of a leaf-litter and soil invertebrate forager. They are opportunistic predators of reptile eggs and small reptiles and have a close association with the celebrated Giant Tortoises of Aldabra, from which they glean insects.

Project justification

The 1998 Aldabra Management Plan established a zoning policy for the atoll which restricts tourist shore excursions to Picard island where the research station is situated. It was decided by the Board of Trustees that re-introducing rails to Picard would not only give added protection to an existing population located on one single island, but would reduce tourism pressure to view rails in restricted areas.

An earlier ill-conceived re-introduction of two rails, by an untrained warden, led to the loss of one bird and the survival of the second bird on its own until it was returned to Malabar. Opinion was divided within the Seychelles Islands Foundation as to the wisdom of re-introducing "a few more rails" or, alternatively, following a scientific approach. In December 1997 at its Annual General Meeting, the Seychelles Islands Foundation decided to approach the Avian Demography Unit at the University of Cape Town to undertake the necessary research into the feasibility of a full re-introduction program. Part of the project was to include a trial translocation of a small but viable population of rails to Picard. Funding for the project was obtained from the Dutch Trust Fund with some counterpart funding by the Seychelles Islands Foundation.

Re-introduction

Similarities between the islands of Picard and Malabar (the rails stronghold) is well established. Picard, containing extensive tracts of *Pemphis* and dense, mixed-scrub habitat, is the obvious location for re-introducing rails. Since the first reliable population estimate was made in the 1970's there has been no significant departure from the estimated 8,000 individuals. This large, stable population afforded the opportunity to undertake a cautious and carefully-measured re-introduction, preempting possible disasters.

Genetics

Geological evidence suggests that the land-rim of Aldabra was breached around 5,000 years ago. The resultant central lagoon fringed by four islands, each separated by deep channels, created a practically insurmountable obstacle to flightless birds. It is thus likely that the rail populations have been isolated for a considerable period and there is some evidence of inter-island morphological differences. Without a prior knowledge of genetic differences between islands, it was imperative that mixing stocks be avoided: only birds from the large population on Malabar were re-introduced. Another genetic consideration was obtaining a representative sample of the genetic diversity of birds on Malabar. To this end, 20 individuals were trapped from the extreme ends of Malabar.

Capture

Rails are inquisitive and opportunistic foragers making trapping a relatively straight-forward procedure. Birds were captured at two locations (Gionnet and Middle Camp) on Malabar, approximately two weeks apart. A simple treadle-release trap, baited with rock crab was used to capture rails. Rails were weighed at capture, blood collected and then housed in crates until they could be transported to Picard. The first group of five pairs were captured from the Gionnet area, and daily trips back to Picard ensured that birds were not held in close confines for more than a day. Owing to the vast distances and tidal restrictions on traveling between Picard and Middle Camp, birds caught there were confined for up to two nights before being moved. Pairs of rails were released into enclosures of approximately 30 m^2, situated behind the beach-crest at the extreme northern end of Picard station and in the abandoned settlement. They were provided with fresh and sea water and fed twice daily on a variety of foods, including rice, fish, fly maggots and shore crabs.

Release

At the time of release (after 7-14 days in captivity), rails were caught, weighed and blood taken for later serological analyses. They all experienced substantial weight-gain during captivity and appeared to be in excellent health. Doors to the enclosures were left open and rails allowed to leave the cages at will.

The second group of birds (from Middle Camp) were brought across almost immediately after the release of Gionnet-captured birds. It was thus possible to monitor released birds which may have wanted to continue to be fed at the cages. One of the five pairs remained in the immediate vicinity of the cages, and they invariably appeared when the whistle to signal feeding time was given. This pair was fed for three months; their reliance on supplementary food tailed-off after the onset of the rainy season in late December and ceased early in January 2000.

The rail enclosures are bounded by two paths, approximately 70 m apart, between which lies classic rail habitat. After release, an unpaired individual and five pairs established territories around the two paths. These birds could be tracked, despite the dense scrub, by call-playback of recorded rail vocalizations and whistled imitations of their duetting song. One of these pairs consisted of birds not paired when released; the other four were original pairings which remained intact. Three unpaired individuals were seen occasionally after release and five birds have not been seen since their release. A sixth pair was heard duetting deep in impenetrable *Pemphis* approximately two months after release. On 24th January 2000, one pair had produced three chicks.

Problems

The death of two birds in transit is the only setback of an otherwise successful re-introduction. An unfortunate combination of events conspired to result in the death of the birds; the single greatest factor can undoubtedly be attributed to the incorrect sexing of one bird. The two birds were found foraging approximately 2 m apart; one bird showed bill-characteristics used in the field to identify females. They were captured and kept in a crate for two nights, before being transported from Middle Camp to Picard. The birds appeared weak before being loaded on the boat, by the time they were placed in their enclosure on Picard several hours later, they were utterly exhausted. They died some time that night. Post-mortem revealed both birds to be males. It is assumed that the stress of capture, combined with a lengthy confinement in close quarters with another male proved too much. While the combination of events leading to their death was unfortunate, it was also most informative.

Aldabra White-throated Rail
***Dryolimnas cuvieri aldabranus*.**

Future work

Besides the genetic analysis of the re-introduced birds, intensive monitoring of all breeding activity, post-release dispersal, body condition and other behavior has been maintained. Further, a full compliment of around 50 individuals will be re-introduced in the near future, to ensure against founder effects and stochastic events to which small populations are so prone.

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Use of mathematical models in bird re-introductions—some examples from the Seychelles

Justin Gerlach

Introduction

Mathematical models of population dynamics have been available for many years, and although they have been examined in detail from theoretical and ecological viewpoints, they have rarely been used in re-introduction planning. In theory, they provide the possibility of pre-release prediction of the success or failure of re-introductions and evaluating subsequent progress. There are many population dynamics publications which are generally highly technical and this can be a discouraging factor in their use by re-introduction planners and practitioners. However, relatively simple mathematical models can be devised that reflect known population processes very accurately.

Three examples from the Seychelles islands are summarized below to show the potential application of models to re-introduction planning. These models are the subject of several papers in preparation and are not described in detail but brief summaries of their predictive powers are given.

Seychelles magpie robin

The re-introduction of the Seychelles magpie robin *Copsychus seychellarum* to Cousin, Cousine and Aride islands was preceded by research into the wild ecology on Fregate island. This research has resulted in the accumulation of extensive data on population processes in the species. These data have been combined into a highly accurate model that describes the changes in the Fregate population over the last 30 years. This model has been applied to the re-introduced populations on other islands and the predictive power is repeated (Fig. 1).

Seychelles magpie robin
Copsychus seychellarum

For Cousin and Cousine islands the model predicts significant population growth with future declines due to social factors and demographic instability in these small populations. The problems of demographic instability have already been noted (Gerlach & Le Maitre, 2000) and available data suggest that these small populations are not viable in the long-term. Application of the model to the re-introductions on Aride island predict repeated failure to establish more than a single bird, as has been the case since 1978. The latest release on that island is predicted to result in a non-viable population of 1–3 individuals. The Seychelles magpie robin model provides an indication of the long-term suitability of selected islands. Initial rapid population growth is not a reliable indication of long–term success and the population dynamics of this particular species suggest that the use of small islands may be problematic in the long–term.

Seychelles warbler

A noted contrast is the Seychelles warbler *Acrocephalus seychellensis* where a model predicts explosive population growth followed by stabilization (Fig. 2). In this case island area is less significant and very large populations are expected to be maintained. As with the Seychelles magpie robin the model predictions are borne out by reality. Here model data support the assumptions of the re-introduction and could be extended to other islands to predict the probability of re-introduction success.

Fig. 1. Prediction model for Seychelles magpie robins

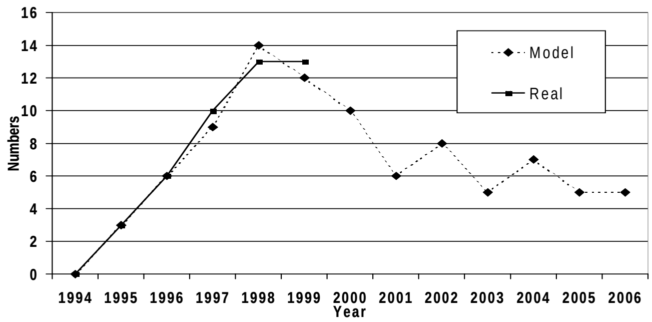
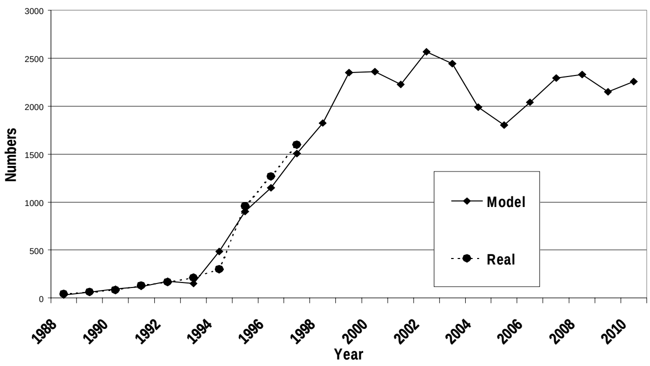


Fig. 2. Prediction model for the Seychelles warbler



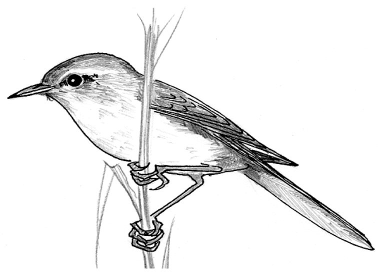
Aldabra white-throated rail

For both of the above examples the models were developed after the re-introductions. For the Aldabra white-throated rail *Dryolimnas cuvieri* the experimental nature of the re-introduction to Picard island allowed a model to be developed and tested in tandem with the re-introduction project.

Preliminary data allow only a basic model to be developed. This provides a reasonable correspondence with the source population, predicting a stable population of some 7,500 birds. This suggests that the model will provide at least a preliminary estimate of the likely population development on Picard Island. This preliminary model predicts a relatively gradual population growth to a stable level of some 2,000 birds. As the re-introduction progresses new data will be added into the model to refine its predictions and census data will be compared with the predictions to determine whether or not any significant overlooked mortality factors are suppressing population growth.

Conclusion

These models provide a means of evaluating the success of re-introductions and a prediction of possible future trends. Without comparative data or model predictions it is difficult to provide an objective assessment of re-introduction success or to identify approaches that would have yielded more stable populations. Models may endorse existing programs, or they may provide evidence of dynamic problems that may be obscured by initial rapid population growth. If they are integrated with the re-introduction program at an early stage they provide a means of predicting changes, evaluating success and identifying important mortality factors that may be obscured by ecological complexity. In all re-introductions, particularly where re-introductions are constrained to work with small islands with great ecological instability, population modeling should be an invaluable tool for the future.



Seychelles warbler
Acrocephalus seychellensis

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The South African wattled crane supplementation program

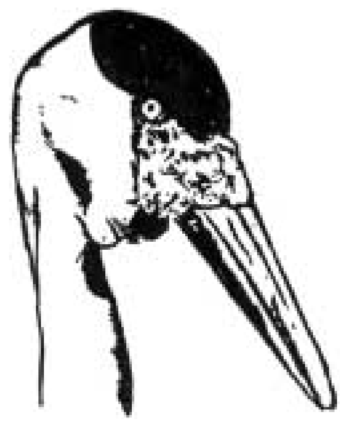
Ann Burke & Lindy Rodwell

The South African population of wattled crane *Bugeranus carunculatus* is estimated at 250 individuals and is classified as critically endangered under IUCN Red List Criteria. South African wattled cranes are most threatened by habitat destruction and degradation, collision with utility lines, disturbance at breeding sites and accidental and purposeful poisoning. Field conservation programs have been actively addressing the threats to wattled cranes since 1985, and a variety of research studies and monitoring programs have been undertaken over the last 20 years (Meine & Archibald, 1996).

In 1994, the Southern African Crane Working Group was established and became a working group of the Endangered Wildlife Trust in 1996. In 1995, a "Crane Habitat and Management Plan" was written to prioritize national crane conservation efforts. The plan identifies four main program initiatives: 1) Education and awareness, 2) Habitat protection, 3) Research and monitoring, and 4) Captive breeding and supplementation (Prospectus of the South African Crane Working Group, 2000). Southern African Crane Working Group directs and assists established crane groups and field officers located in the seven key crane regions in South Africa. It also serves as a coordinating body for crane conservation projects, wetland and grassland experts, government officials and international crane experts.

Education and awareness programs target principal "crane custodians" such as landowners, farm workers, local communities, and rural and urban schools. Multi–faceted environmental awareness campaigns are being used to focus attention on wetlands, cranes, the responsible use of water and the correct application and storage methods for agrochemicals. Habitat protection measures include inventory of wetlands and wattled crane breeding/feeding/roost sites, power line marking, and lobbying for legislation to improve wetland protection and provide stricter control of forestry and agricultural permits. A national color ringing program, blood collection for genetic and toxicological studies, research into habitat use, aerial and ground censuses and a captive-breeding program are other important aspects of the national wattled crane conservation program.

In 1993, the regionally based Highlands Crane Group, the South African Crane Foundation and Southern African Crane Working Group formulated plans for a wattled crane release program based on IUCN Guidelines for Re-introduction in the Verloren Valei Nature Reserve in the province of Mpumalanga near Dullstroom, was selected as the rearing and release site. This 6000 ha reserve was established in 1982 for the protection of nesting wattled cranes and in February 2000 was assigned Ramsar site status. It contains high quality habitat secure from human disturbance. The program works in full cooperation with the national government and the Mpumalanga (provincial) Parks Board. For three years prior to the release, the Highlands Crane Group conducted community and education awareness programs in the vicinity of Verloren Valei Nature Reserve. A Catchment Management Plan was developed to guide wise use of water resources in the area. Adequate funding was secured for all phases of the wattled crane supplementation program. Total program costs are approximately US$ 10,000/year.



Wattled Crane
Bugeranus carunculatus

Release stock comes from second eggs collected from wild nests. Second egg collections have a minimal impact on wild wattled crane productivity as, 1) wattled cranes only rear a single young per breeding attempt and 2) are only conducted after the viability of the first egg has been established. Southern African Crane Working Group is utilizing a chick rearing technique that has proven successful in the whooping crane *Grus americana* and Mississippi sandhill crane *Grus canadensis pulla* release programs. This technique is known as "isolation" or "costume" rearing. The technique allows a large number of chicks to be reared at one time, ensures proper imprinting and through human avoidance conditioning, produces chicks fearful of humans (Nagendran *et al.*, 1996). The chicks are supervised by a costumed human and from approximately 3–4 months of age are housed in a large, predator proof wetland pen where natural foraging and roosting behaviors are encouraged.

IUCN Guidelines for Re-introductions state that releases should only occur after a self–sustaining captive population has been established. Although Africa does not have a self-sustaining captive population, the United States does. Through the wattled crane Species Survival Plan (SSP), and the wattled crane Global Animal Survival Plan (GASP), the United States provided South Africa with six adult birds (1996) and eight eggs (1999) that were genetically surplus to the U.S. flock (Beall, 1996). These birds, along with four isolation reared chicks from the release program, have been used to increase the number of captive breeding stock with the goal of establishing a self-sustaining flock in South Africa. Southern African Crane Working Group has promoted the concept of managing all captive birds as one flock based on genetic goals and has developed specific terms and conditions for the seven facilities currently holding a total of 24 wattled cranes.

Prior to 1995, three main populations of wattled cranes were recognized. The species is most abundant in south central Africa (southern Zambia, Botswana and Mozambique) with smaller populations occurring in Ethiopia and South Africa. Within this range, the Ethiopian population was thought to have the greatest potential for genetic distinctness. However, consistent with South Africa signing the Convention on Biodiversity in 1995, the need for a taxonomic assessment of the South Africa crane population was identified. In 1999, a comprehensive genetic/blood analysis study was begun on the three crane species in South Africa. Blood samples from Zimbabwean and Botswanan wattled cranes are also being included in the analysis. If the South African wattled crane proves to be significantly genetically distinct from wattled cranes in other parts of its range, steps will be taken to ensure that only genetically suitable stock is being bred and released. Table 1 summarizes the supplementation program between 1995 and 1999.

Prior to the release, health testing and quarantine procedures are practiced. In 1995, a four-month old color ringed, radio tagged chick was released on the Verloren Valei Nature Reserve. Three months after release the bird was killed by a caracal *Felis caracal*. Post–release monitoring revealed that the chick was choosing upland habitat to roost rather than open water. The whooping crane re-introduction program has found that cohort size influences post release behavior and that groups of 6–8 birds appear to be the optimal size to persist as a cohesive unit following release. Efforts are now being made to collect and rear a minimum of 5–6 wattled crane chicks per season.

In 1997, the five birds were released as a cohort. They were between 4-8 months of age. Upon release, the birds chose to roost at night in open water wetlands. The cohort of five survived for over eight months utilizing reserve land and privately owned farmland. The cohort then made an unexpected dispersal movement approximately 100 km southwest of Verloren Valei Nature Reserve. This area had not been covered with any education/awareness field program and it is not within the historical range of the wattled crane. On 16th October 1998, two birds were killed by collisions with power lines. Three weeks later, the other three died after ingesting poisoned grain. Persons had illegally treated the grain with monocrotophos (an organophosphate) most likely in an effort to capture game birds (i.e. Helmeted Guineafowl *Numida meleagris)* for food.

Southern African Crane Working Group did not feel these deaths were in vain as the Dullstroom community was highly upset that "their" wattled cranes had been killed. The event further served to educate people on the threats confronting wild cranes. Within a day of the power line collisions, ESKOM, the South African Utility Company, marked 300 meters of power lines with Bird Flappers (line marking devices) where the incident had occurred. In additional, the Poison Working Group of the Endangered Wildlife Trust hired a full–time field coordinator in 1999 to encourage the responsible use of agrochemicals in the area.

In 1998, four chicks were released. Seven weeks following release one was killed by a caracal and three are still surviving on private farmland adjacent to Verloren Valei Nature Reserve. Over the first three years of the program, predator losses account for 20% of the mortality.

In 1999, a Wattled Crane Recovery Team was established. The team is composed of persons directly involved with the second egg collections and chick rearing, the Verloren Valei Nature Reserve manager, captive breeding specialists, a media coordinator, field officers working in the release area, the National Crane Conservation Project Coordinator and the Co-chair of Southern African Crane Working Group. The team meets on a regular basis to review all aspects of the release project. Through the evaluation process, it was recognized that association with wild conspecifics improved survival rates of released whooping and sandhill cranes *Grus canadensis*. In 2000, the Wattled Crane Recovery Team decided to experiment with the one-by-one release technique pioneered by Dr. David Ellis in the United States and release isolation reared wattled crane chicks one at a time into the non-breeding flock (Ellis *et al.*, in press).

In February 2000, an 8.5–month-old female was released to the non–breeding flock of 36 adult birds on a privately owned farm protected as a Natural Heritage Site. This female was the most dominant member of the 1999 cohort. Approximately two hours following release, the bird encountered a group of 11 adults that showed extreme aggression towards her. Over the following 10

Table. 1. Summary of the South African wattled crane reinforcement/ supplementation program: 1995–1999

YEAR	NUMBER OF EGGS COLLECTED FROM THE WILD	NUMBER OF FLEDGED CHICKS	FATE
1995	1	1	Released, killed by predator
1996	2	2	Kept in captivity for breeding purposes
1997	3, (2 additional eggs produced in captivity)	5	5 released, 2 killed by powerline collisions, 3 killed by poisoning
1998	10 (2 infertile)	5	1 kept in captivity, 4 released, 1 killed by predator, 3 surviving
1999	5	4	1 kept in captivity, 3 released, 3 surviving

days, she frequently foraged in close proximity to the wild flock, but she stayed in the upland habitat 24 hours per day. On the 11th day, the bird followed the flock and roosted with them in an open water wetland. Two months following release, the bird has formed a close association with a similar-aged bird and continues to forage and roost with the wild flock. On 12th April, 2 additional 9.5-month old birds (male and female) were released as a cohort to the remaining wild flock of 19 birds. Two hours following release, the female moved from the farm followed by the male 30 hours later. Both birds flew less than 1 km onto adjacent, privately owned farmlands. Both were still alive at the time of this writing.

A 'premature' review of the current releases seems to indicate that wattled cranes released to a flock of con–specifics, using the one–by–one technique may be the way forward. This experiment has however revealed that under current management, birds released using the one–by–one technique are not exhibiting appropriate roosting behaviors immediately after release. Plans are underway to create open water areas within the holding runs where the chicks are housed at night. The program also plans to expose the chicks to a wattled crane decoy. This decoy technique has been successfully used in the U.S. as it ensures proper roosting behavior and gives managers a degree of control over the birds after release. A set of behavioral and physical criteria that each juvenile must meet in order to be considered a candidate for release is currently being developed by the Wattled Crane Recovery Team. If an individual does not meet the specified criteria, it will be kept as captive stock.

The one–by-one release experiment also has shed light on the gaps in our knowledge of wattled crane social structure in non-breeding flocks and dispersal movements from the time juveniles leave their parents to when the birds become established as breeding pairs. To further analyze conservation efforts to date and better clarify future release and research efforts, a wattled crane Population and Habitat Viability Analysis workshop is scheduled to be held 31st July–3rd August 2000 in Wakkerstroom, South Africa.

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Conservation of the pink pigeon in Mauritius

Kirsty Swinnerton et al.

Introduction

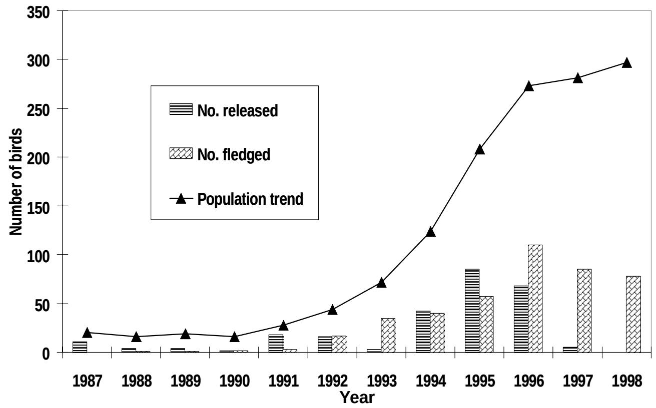
The Pink Pigeon *Columba mayeri* is endemic to the island of Mauritius in the Indian Ocean. Its has been rare for nearly a century and by the early 1980's only 10-20 individuals survived in the wild. A conservation program for the Pink Pigeon was started in the mid-1970's but it was not until 1987 that releases into native forest began (Jones *et al.*, 1992). A re-introduction program was carried out between 1987-1997 alongside management of the remaining wild population. The following article has focused on re-introduction and management techniques which were particularly relevant to the population recovery.

During the first four years of the program, the population declined from 20 to 16 birds in 1990, despite 21 birds released and four fledged in the wild (Fig. 1). The population started to grow in 1991 and increased on average by 49.5% per year up to 1995. From 1996, population growth slowed and increased by 10.5% per year to reach a peak population size of 297 birds at the end of 1998 in four sub-populations; Brise Fer (79), Bel Ombre (81), Ile aux Aigrettes (60) and the remaining wild population at Pigeon Wood (77). Up to 1998, 429 birds had fledged in the wild, 77% of which fledged between 1995 to 1998 and the population is currently about 71% wild-fledged birds.

Re-introduction into native forest

During the re-introduction program, 256 birds were released at two sites in the Black River Gorges, Plaine Lievre and Bel Ombre, and at a third site, Ile aux Aigrettes, a 25 ha offshore island nature reserve. Most birds were released in a three year period between 1994 and 1996 when 76% of all birds were released. Releases were stopped in January 1997 owing to a greater number of birds fledging in the wild. Of the 256 birds released 239 were captive-bred and 17 were rescued from the wild and captive-reared. Most release birds were captive-bred on Mauritius at the Gerald Durrell Endemic Wildlife Sanctuary but five birds were bred at the Jersey Zoo (Table 1).

Figure 1. Population trends of pink pigeons on Mauritius: 1987–1998



Release procedure and technique

Birds were held in an aviary at the release site for about one month prior to release. This allowed them to familiarize themselves with their surroundings, with already established birds and with their keeper. It also established the aviary as the focal point to which the pigeons could return for food. The release aviaries were designed with 4-5 interconnecting compartments so that difficult or timid birds could be separated and enabled a group of birds awaiting release to be held at the same time as a group was undergoing release (Jones *et al.*, 1992). No formal pre-release training was given, but some groups were provided with branches of wild food inside the release aviary.

The first releases at Plaine Lievre in 1987 followed the procedure and techniques developed at Pamplemousses Botanic Gardens (Todd, 1984), but as releases progressed refinements were made and 'softer' release methods were developed (Jones *et al.*, 1992). Birds were released when hungry and food was provided immediately on release which encouraged them to stay close to the release aviary. Pigeons were first released about one hour before dusk and only in calm weather. This ensured that birds did not wander far from the aviary and could easily relocate it the next morning for food.

During the earlier releases (up to February 1989), the birds were left at liberty after the first release and given food daily at the aviary. Since then, after release birds were re-trapped just before dusk the same day or at the earliest time the following day. Typically the birds came out and fed, flew to a nearby tree or re-entered the aviary. When birds roosted outside the aviary, they were re-trapped about 1-2 hours after dawn the following morning. On subsequent days the birds were released for an increasing length of time, bringing the release time forward and re-trapping the birds later. This allowed the birds to familiarize themselves with their surroundings and their sources of food. It enabled the birds to be re-trapped during periods of adverse weather, and minimized the risk of losing birds which panicked and flew out of the area. Birds were usually given full liberty after about one month of re-trapping and releasing. As the population established around the release site it became easier to release further birds into the population and birds were allowed full liberty earlier.

Pigeons were typically released in groups of 4-6 birds (Table 1). Early on in the program, surplus older captive birds were used to test release techniques. However, older birds proved problematic and juveniles were favored in all releases from July 1991. The median age at release was 86 days (sd=653, n=265).

Post-release monitoring

All birds were identified by a unique color and numbered ring combination fitted to the legs. Radio-tags were fitted to 84 (31%) birds and hawk bells fitted to 74 (29%) birds for monitoring post-release. Usually at least half the release group had a bell or radio-tag fitted. Hawk bells were preferred later in the program as they were cheaper than radio-tags. Radio-tags and bells were removed once the bird gained full-liberty.

Management of birds in the wild

Supplementary feeding: Supplementary feeding was considered essential to the survival of newly released and established birds. Feeding platforms were built close to the release aviary and within the breeding grounds when breeding birds were established. The platforms were protected from rats and a mix of cracked maize and wheat was permanently available from a hopper on the platform. The food sustained birds through periods of food shortage and improved productivity by improving the condition of breeding birds, increasing squab and fledgling survival and reducing fledgling dependence, which subsequently reduced recycling periods.

Predator control: Introduced mongooses *Herpestes auropunctatus* and cats *Felis catus* were controlled at release sites and within breeding grounds to improve survival of pigeons. Box traps were spaced on a grid throughout the pigeons breeding area, which was surrounded by a peripheral ring of traps. Traps were at higher densities around the release aviary and supplementary feeding stations, and additional traps were placed where field signs of predators were found. The strategy was to create a core predator free area and to catch immigrant animals.

Rats *Rattus rattus* were controlled using an anti-coagulant poison Brodifacoum supplied in commercial wax blocks. Rats were controlled within the breeding grounds to reduce depredation of eggs and young squabs, and around the release aviary to reduce food spoilage. Poison was placed in bait stations on a 50m grid which was surrounded by peripheral stations 25m apart.

Disease research and control: From 1987-1993, cloacal and buccal swabs were taken from birds prior to release and screened for bacteria and parasites. From 1993, a more general approach to disease research was adopted to understand the risks the species faced in the wild and how it affected their survival. Birds were screened for blood parasites, specifically *Leucocytozoon marchouxi*, avian pox, faecal parasites and trichomoniasis. More recently, birds were screened for herpesvirus, avian reovirus, avian adenovirus and Paramyxovirus with no positive results. About 30% of wild and free-living pigeons were infected with *Leucocytozoon marchouxi* which can be fatal to young birds. Trichomoniasis, together with infection of *Leucocytozoon marchouxi*, severely affected squab and fledgling survival on Ile aux Aigrettes but not at mainland sites.

Table 1. Summary of pink pigeon releases in native forest on Mauritius			
	Brise Fer	Lie aux Aigrettes	Bel Ombre
Release period	1987-97	1994-9	1994-97
Number of groups released	31	16	17
Number of captive-birds released	107	46	88
Number of wild birds released	1	9	7
Number Males:Females:Unknown Sex	51:37:20	27:22:06	35:43:17
Mean size of release group (>1)	4	6	6
± sd (birds)	2.08	2.85	1.52
Number of single birds released	2	8	0
Median age at release	88	171	72
Number of birds fitted with radio-tag bell	56:26	14:7	14:41
Number survived at 30-days post release	94	52	84
% survived at 30 days post-release	87	95	88

Pink pigeon *Columba mayeri*

Reservoirs of infection, mainly introduced pigeons and doves, were controlled at feeding stations and hoppers were designed to exclude introduced species. On Ile aux Aigrettes, the high incidence of trichomoniasis was likely caused by water shortages at certain times of the year. Water, provided in drinking hoppers, was changed daily and treated with chlorine. The pigeons were treated with an anthelmintic in the drinking water two or three times a year to reduce the parasite load in the population. Fledglings and squabs monitored on the nest were treated individually but recently, trials monitoring the survival of treated and untreated squabs were undertaken.

Egg and brood manipulations: Nests were manipulated to improve breeding success and increase the number of fledged young, which was particularly important early in the program. Techniques included harvesting eggs for captivity, transferring eggs between nests, fostering squabs between wild nests, fostering squabs from captivity to wild nests and rescuing squabs from the wild.

Eggs were mainly harvested early in the program from the remaining wild population in Pigeon Wood between 1989 to 1993. This was due to high depredation of previous nests by rats and monkeys and many viable eggs were being lost. In addition, by the end of 1989 productivity in captivity had declined owing to the increasing age of breeding females (Jones, 1995) and the release program was severely compromised by the lack of young birds available for release.

Transferring eggs and fostering squabs between nests have the potential to improve productivity by spreading risk and reducing the brood size per nest and provide parents with valuable rearing experience. Eggs were also removed to captivity to reduce losses through depredation and squabs fostered back if incubation was carried through successfully. Rescued squabs were reared in captivity and provided extra birds for release, as well as improving survival for the remaining squabs in the wild.

Rescuing squabs from wild nests and harvesting eggs produced the most number of young per nesting attempt, although none of the progeny from harvested eggs were returned to the wild. The success of fostering attempts was likely compromised by poor squab survival, due to disease, on Ile aux Aigrettes where most of the manipulative techniques were carried out. All manipulations involved intensive monitoring and management of the eggs and squabs and daily inspection of the nest. Nest manipulation was easiest on Ile aux Aigrettes where the shorter nest trees were accessible. On the mainland, the greater risk of damaging the nest during access restricted the number of manipulations being made.

Discussion

Survival of released birds up to 30 days post-release was 89%. Survival to breeding age (year one) was 75% and mean adult survival (years 1-6) was 81% per year (±0.06%). Some released birds survived for many years, the oldest surviving captive-bred bird released as a juvenile was a male released in 1987 who died at 8.4 years old. The oldest surviving captive-bred female released as a juvenile was 7.2 years old in 1998 when she was still alive.

Several key elements of the re-introduction and management program can be identified which contributed most to the recovery of the pink pigeon population. The development of soft-release methods and post-release support of birds minimized losses during the release process. Large numbers of birds available for release was considered one of the main elements of its success. In the first four years of the project only 21 birds were available for release, an average of about five birds per year, and the released population actually declined from eight to six birds in 1990. Releasing small numbers of birds made it hard to establish a group. Birds were established more quickly once a core group of birds existed. Few birds meant that experimentation with release techniques was limited. The release program was continuously reviewed as it progressed, techniques were refined according to failures or successes and management techniques were developed or modified as the population increased. In-country captive-breeding was considered necessary to provide birds for release. The disease risks and logistical and bureaucratic problems of importing birds from captive populations outside of Mauritius restricted the number of imported birds available to the release program.

Juvenile birds were easier to release than mature adults as they associated with established birds learning from them about their environment. Released adults would fight or compete for mates with territorial birds and often be driven out of the area. Egg and brood manipulations were important early in the program when just a few fledglings contributed to population growth but the effort involved, particularly on the mainland, do not make them a long-term management option. Problems associated with disease on Ile aux Aigrettes, the only lowland site, were not anticipated and may have an important influence when deciding future release sites. However, disease resistance should be encouraged by only controlling disease where it severely limits the population.

Supplemental feeding and predator control for newly released and established birds is considered essential. Studies show that survival and productivity are improved with supplemental feeding and predator control and enables birds to utilize degraded and marginal habitats. In addition, birds can exist at higher densities than the habitat would normally support and thus help to maintain a viable population. To ensure long-term sustainability of the population it is intended to manage core areas of breeding birds and their habitat.

Latest releases

The re-introduction of the pink pigeon to Combo, the newest field site within the Black River Gorges National Park, has now been ongoing for a year. It has been a tremendous success with the new sub-population quickly being established and both survivorship and breeding success have been exceptionally high. Since 7th May 1999, there have been eight groups of between 3-6 pigeons released. A ninth group is currently being released. This will bring the total number of pigeons released at Combo to 45. Of these, 31 were captive bred at the aviary facilities in Black River, and 14 young birds were caught at other pigeon sub-populations and translocated to Combo. To date, these released birds have reared eight fledglings to independence. Forty-one pink pigeons are now regularly seen at Combo. In addition, eight pigeons have dispersed from Combo to the other sub-populations in the Black River Gorges National Park, which improves the spread of genetic material throughout the population. With the successful establishment of this sub-population and a productive 1999/2000 breeding season there are now over 400 free-living pink pigeons on Mauritius.

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The release of captive bred echo parakeets to the wild, Mauritius

Lance Woolaver et al.

Introduction

The echo parakeet *Psittacula eques echo* was once considered to be the rarest parrot in the world. In 1986, the population was estimated at only eight to 12 individuals (Jones, 1987). Conservation efforts to recover the parakeet began as early as 1973 with the work intensifying in 1987. Many techniques have been attempted in the past including habitat protection and restoration using fenced and weeded forest plots or Conservation Management Areas, predator control around nest sites, manipulation of breeding pairs (double clutching and chick fostering), supplementary feeding and provision of nest boxes (Jones & Duffy, 1993). Between 1993 and 1999 the program was further refined. The main emphasis is now on clutch manipulations and downsizing of wild broods, regular examination of active nests and weighing of chicks, predator control and nest cavity improvement and the release of young captive raised parakeets back into the wild. The echo parakeet is one of the most intensively managed avian species in the world.

The main limiting factor for the species is low survival of young chicks in the wild. Due to the degradation of forest there is a limit to the amount of food available during the breeding season. A number of secondary factors are found to affect survival of nestlings, including nest cavity competition with exotic avian species, infestations by nest fly larvae *Passeromia heterochaeta*, and predation by introduced mammals. These secondary factors are not likely to be as much of a problem with a healthy wild population but are important while the population remains low.

As of March 2000, there was a minimum estimated population of 126 echo parakeets, 20 of which were in captivity at the Gerald Durrell Endemic Wildlife Sanctuary in Black River, Mauritius. During the 2000/2001 breeding season there will be 26 females of breeding age in the wild. This is primarily due to management techniques which have shown great success in the last few seasons. The overall goal of the recovery program is to produce the maximum number of high quality fledglings in the wild each season. The ability to downsize nests (leaving each wild nesting pair with a single chick) has greatly increased the survivability of the wild fledglings. In some cases, experienced pairs have been left with two chicks. Previous to downsizing, most nests failed as parents attempted to feed all chicks. The number of fledglings produced in the wild has increased dramatically since 1996 when three chicks fledged from wild nests. This was followed by seven in 1997, 11 in 1998 and 18 fledglings from the wild during the 1999/2000 season "extra" chicks which are taken from wild nests are either fostered to other wild nests which have failed, or are taken down to the captive breeding center in Black River. Since 1997, 22 of these captive raised chicks have been released back into the wild.

Release techniques

Release techniques had been fine tuned during a trial release of Indian ring-neck parakeets (Murray, 1998). From the trial release it was found that parakeets should be released in small groups of two to five young birds which are trained to associate a whistle upon the presentation of food. The use of falconry bells for locating release birds during the first few weeks of release was invaluable. The trial release also stressed the need for an acclimatization period of 2 to 3 weeks before birds were let out of the release aviaries. Table 1 shows the numbers of echo parakeets released between 1997 and 1999.

In 1997, the first echo parakeets were released. This first group consisted of two males and a female. Three males and eight females were released in 1998. Six males and two females were released in 1999. Of these, five males and eight females are known to still be alive. Two of these were bred in captivity with the rest being originally taken from wild nests, either as eggs or very young nestlings.

Nestlings are taken from the nest when around 10 days of age. This allows them to have had exposure to gut flora from their parents and these older chicks show greater immune abilities than younger chicks. We prefer to downsize chicks, removing them from the wild while still healthy and leaving the parents with fewer mouths to feed, than to wait until chicks are showing signs of starvation or dehydration. Chick mortality was much higher during the earlier, learning years of this project when chicks were taken from nests after they had already been compromised. After arrival at the aviaries, chicks are captive raised and are either kept at the aviaries to increase the number of breeding pairs in captivity or are subsequently released back into the wild after they have been successfully weaned.

Table 1. Number of echo parakeets released/year

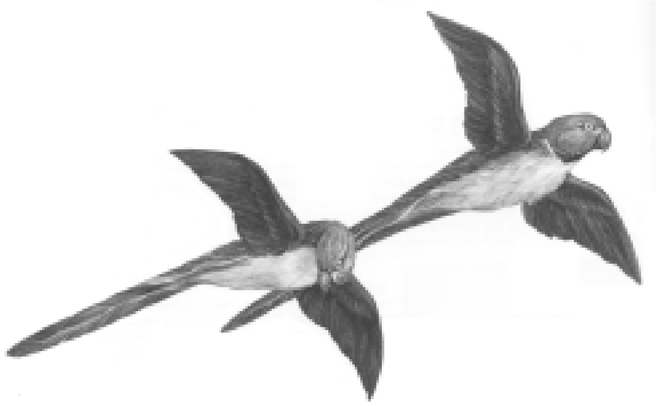
GROUP	DATE OF FIRST RELEASE	MALES	FEMALES
1	03/08/97	2	1
2	04/03/98	0	2
3	27/03/98	1	3
4	30/04/98	2	3
5	27/02/99	3	1
6	03/04/99	3	1
	TOTAL	11	11

Echo parakeet *Psittacula eques echo*

Echoes have been captive raised at the Gerald Durrell Endemic Wildlife Sanctuary through a combination of hand-raising and of foster raising by ring-neck parakeets and echo parakeets. Echo parakeets raised by ring-necks have never been considered as potential release birds. Primarily hand-raised birds have been used as it has been important that the first release birds be comfortable around staff. In future releases we will be releasing birds which have been raised by our captive pairs of echo parakeets. Any future hand-raised birds will be brought up to the release aviaries at a younger age where they can then be weaned. This will allow them greater contact with the established birds in the forest around the release aviaries. Established birds show a great deal of interest in release groups and interact with them through the mesh of the aviaries.

The release birds are brought to the release aviaries in the forest when the youngest is 100 days old. They are released in groups of three to four at a time. They are kept in the aviaries for two weeks during which time they become accustomed to the climate and are able to strengthen their flight muscles. They are able to socialize with established release parrots outside the aviaries. They are fed a combination of native and exotic fruits and vegetables and the enclosure is filled with native branches and leaves. Feedings are accompanied by a whistle so that the birds associate the whistle with food. After the two weeks, the soft release program is begun. Hatches are opened in the side of the release aviaries closest to the native forest 15 minutes before dusk. The parakeets hop out into the nearby branches where they roost for the night. The release staff wake up early the next morning before the parakeets are active and call them back into the aviaries with food and its associated whistle. The parakeets are then kept inside the aviaries until the evening. Each day they are let out 15 minutes to a half hour earlier and each morning they are called in a bit later so they are allowed more and more time outside each day. This release process takes another two weeks until the parrots are free. The aviaries are left open and fresh fruit and vegetables are placed inside for two months to provide the parakeets with a food source until they have found wild food sources for themselves. Supplementary feeding hoppers at the release site, in the release aviaries and in the Conservation Management Areas are kept full of complete diet parakeet pellets year round. It is important that the release birds be comfortable with entering the aviaries as it has proven a valuable tool when needing to catch ill or damaged birds for treatment.

All release birds are taught to use a feeding hopper while at the Gerald Durrell Endemic Wildlife Sanctuary. These are small



Echo parakeets *Psittacula eques echo* in flight
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plastic containers with a lid which is easily lifted by the parakeets. They are filled with complete diet parakeet pellets provided by Kaytee and Pretty Bird. These feeding hoppers are placed in the release aviaries, outside the aviaries at the release site and throughout the forest in the Conservation Management Areas. The release parrots are not reliant on these hoppers but they do visit the supplementary food stations each morning and evening. Feeding hoppers are also placed near the nesting cavities of each release group to provide additional food during the nesting season. To date, a single wild parakeet has learned to use the hoppers. This is a wild male which had joined with the original release group and learned to use the hoppers as a juvenile. This male has since paired with one of the release females and is regularly seen using the hoppers. In addition to their role as supplementary feeders these hoppers provide staff with sites where they can observe the release birds without having to follow them throughout the forest.

The established release birds have been invaluable in teaching subsequent release birds which means that newly released birds can be less reliant on release staff. Established birds teach new releases which wild foods to eat, how to orientate themselves in the nearby forest and predator avoidance. There is a fine line between release parakeets which are comfortable with humans and those which are over dependant on human company. We have found that over-dependence on release staff has caused behavioral problems in one of the released parakeets. One of the first released males was purposely kept tame. This individual paired with a released female and produced two healthy chicks in September of 1999. He was overly aggressive to his chicks, both of which subsequently died from wounds inflicted by the male. He is the only release parakeets which has shown this aggression. All other release birds have been exemplary parents. Our goal is to release female parakeets which have been raised by captive echo parakeets and for these parakeets to associate the release site, and not the staff, with food. The presence of established release parakeets will help to successfully reintegrate the newly released birds back into the wild population.

Breeding success

The first breeding success came in 1998/99 when the female from the first release group paired with a wild male and laid a single egg. The egg was infertile but a six day old chick from another pair was fostered and accepted. The nestling fledged and has survived its first year and is seen regularly in the area. Four of the six release females of breeding age in 1999 had paired with wild males. Two females had paired with release males. All six pairs have shown signs of breeding activity (nest prospecting and copulating). Two nest attempts failed due to chick mortality while a third female laid an infertile egg. This egg was replaced with a three day old chick which has subsequently fledged and has been regularly observed being fed in the forest by its foster parents. Next season we will have eight release females of breeding age.

Young release parrots show a great deal of interest in the established release and wild birds. Females in particular show a great deal of interest, even in their first year, in wild pairs and wild nest cavities. Within one month of her release in 1999, a young female had left the release site and joined a wild group which was established in one of the fenced plots at Mare Longue. She rarely returns to the release site even though it is only 1 km away. She has shown a great deal of interest in the nest cavity and the nesting activity of the wild pair. This appears to be a common trend in the release females and shows great promise for future releases of females.

Future plans for improvement

The exact cause of mortality is known for two of the nine missing birds. One was killed by a feral cat and the other died of disease. One of the released females was found for sale at a local market. This female was rescued and successfully released again into the forest and she attempted to breed last season. The first release group was attacked by a mongoose but all three birds escaped. Another of the released birds was found to have all of its tail feathers pulled out which could only have been done by a monkey. It is felt that most of the missing birds are likely to have died and we suspect that monkeys and other predators are responsible for many of these deaths. It is possible that some birds have found themselves near the edges of the forest and been the victims of their own curiosity and trust in humans. We hope to address this problem in the future by releasing parent reared rather than hand-raised parakeets. We are not yet sure how to deal with the naiveté of release birds toward predators. At the moment we are relying on newly released birds learning from established birds through alarm calls when predators are around. We are also hoping to fit our released birds with radio transmitters so we can better follow their movements and determine their fates if they go missing. We are continually trying to find ways to reduce this mortality and are very encouraged by the number of birds which have survived and are already contributing to the breeding population in the wild.

The wild population has a sex ratio of 3:1. We are concerned with the fact that there may be a large proportion of males in the wild which are not breeding. Many of these males are likely to be carrying genes which are poorly represented in the current gene pool. For this reason it is important that we try to address this sex ratio imbalance as quickly as possible. The release program allows us to address this problem by only releasing females in the future. The release program also allows us some measure of control over the genetic management of the wild population by allowing us to ensure that genetically important birds are also released and represented in the wild population. We are also planning to increase the number of release sites over the next few years. This will allow us to extend the range of the echo which is at present localized within an area of 40 km^2. The echo parakeet project has benefited greatly from the advice of several geneticists and from its strong involvement with the International Zoo Veterinary Group.

Although a large number of artificial nest boxes of many types have been placed around wild nest sites in the past, wild echoes have never shown any interest in them. It is hoped that release birds can more easily be enticed into using them. A priority of the release program this season is to place wooden nest boxes around the release site and nearby Conservation Management Areas in the hope that release birds will use them and encourage wild birds to use them as well.

The next few years, as our release birds age and become more experienced at breeding we are optimistic that the release program can provide a strong contribution to the recovery of this species. Indeed it already has with the addition of 13 release birds to the current wild population, eight of which are females of breeding age. Two of these have already produced healthy fledglings and we feel optimistic that, with proper management, all eight released females should produce healthy fledglings next season.

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NORTH AMERICA & CARIBBEAN

The Hawaiian endangered bird conservation program

Cyndi Kuehler & Alan Lieberman

The Hawaiian Islands, because of their geographic isolation and evolutionary history, have a highly endemic native avifauna, significant portions of which are quickly being lost to due to the introduction of alien species and disease. The State of Hawai`i, which encompasses only 0.2% of the land area of the United States, has 33% of the nation's total endangered species. The native forest birds remaining probably comprise less than 20% of the original avifauna and without intervention at least three additional species are likely to go extinct within the next 5–10 years. Overall, nearly 75% of the recorded extinctions in the United States have involved endemic Hawaiian species. The limiting factors causing the loss of species in Hawai`i are also responsible for the degradation of native ecosystems. According to recent estimates only around 15-20% of the native ecosystems remain intact.

The Hawaiian Endangered Bird Conservation Program is a unique partnership composed of government agencies (The Department of the Interior and the State of Hawai`i), the Zoological Society of San Diego, the Peregrine Fund and private land–owners working together to develop restoration techniques for 12 species of endangered Hawaiian forest birds. Hands–on recovery strategies are being used to increase reproductive output in rare bird populations during this period of environmental crisis. Wild eggs are collected and artificially incubated and chicks are hand–reared; juveniles are subsequently released or retained in captivity for propagation (Kuehler, *et al.*, in press). To date, over 200 endemic passerines (12 species) have been hatched at the Keauhou Bird Conservation Center and Maui Bird Conservation Center (Table 1). In 1999, this collaborative effort resulted in the first successful passerine conservation program where captive–bred birds (offspring of parents which originated as wild–collected eggs) were re-introduced and subsequently survived and successfully fledged chicks in the wild (Puaiohi). These intervention restoration techniques provide a means to preserve options while the habitat is secured and wild populations are stabilized. However, captive propagation/re-introduction programs are costly endeavors and not the best conservation strategy for all Hawaiian species. The Hawaiian Endangered Bird Conservation program endorses commensurate action to protect and enhance the habitat required to maintain and re–establish viable self–sustaining wild populations of avian species.

Nukupu'u © Gwendolyn O'Connor

Propagation facilities

The construction of the Keauhou Bird Conservation Center on the Big Island of Hawai'i was completed in 2000 and includes an incubation and brooding building with laboratories, fledging aviaries, office space and neo-natal food preparation area. Also, part of the facility are two forest bird buildings with 37 aviaries and bird kitchen, 10 `Alala aviaries, a workshop and two caretakers' accommodations. The operation of the Maui Bird Conservation Center became the responsibility of The Peregrine Fund in March, 1996. This facility has areas for incubation and hand–rearing, 'Alala and Nene Breeding complexes, and indoor-outdoor forest bird aviaries. The facility also serves as an incubation and neonatal area for the endangered Maui forest bird eggs that are brought from the field for captive management.

Captive propagation

Since the inception of the program in 1993, over 200 native Hawaiian forest birds of 12 species have been incubated and hatched. Seven of these species are classified as Federally endangered, to include the `Alala, Maui Parrotbill *Pseudonestor xanthophrys*, Hawai'i Creeper *Oreomystis mana*, 'Akepa *Loxops coccineus*, `Akohekohe *Palmeria dolei*, Puaiohi *Myadestes palmeri*, and Palila *Loxioides bailleus*, and five non-endangered native species; `Oma`o *Myadestes obscurus*, Hawai'i `Elepaio *Chasiempis sandwichensis*, `Apapane *Himatione sanguinea*, `I`iwi *Vestiaria coccinea*, and Common `*Amakihi Hemignathus virens*). These latter species serve as surrogate models for the development of captive propagation and release technology.

'Alala

With the wild population of `Alala numbering less than 10 individuals, the Hawai'i Endangered Bird Conservation Program joined with the `Alala Partnership (Service, and McCandless, Kealia and Kai Malino Ranches, Kamehameha Schools Bishop Estates) in an intensive re-introduction program (Kuehler, *et al.*, 1995). During the period from 1993-1999, 36 'Alala have been hatched and 34 survived to fledging (Kuehler, *et al.*, 1994). 27 'Alala have been released into historical habitat in the South Kona District of Hawai'i. 25 birds survived until independence (~120 days post-release). Although the long–term survivorship of the released 'Alala has been lower than first expected, through the release and monitoring program, biologists have been able to better identify the factors that limit the long-term 'Alala survivorship in the native Hawaiian forests (predators, disease, etc.). In an effort to accelerate the recovery of the 'Alala, the Service is reviewing the options to establish an additional (alternative?) release site with expanded and enhanced habitat

Table 1. Captive–rearing summary: Hawaiian forest birds/Hawaiian endangered bird conservation program: 1993–1999

SPECIES	YEAR	EGGS COLLECTED	VIABLE AT COLLECTION	HATCH	SURVIVE 30 DAYS	% HATCH	% SURVIVE
Common ‘Amahiki *Hemignathus v. wilsoni*	1994	9	6	4	5*	66.7	75.0
	1995	29	20	17	16	85.0	94.1
Common ‘Amahiki *Hemignathus v. wilsoni*	1997	2	0	-	-	-	-
	1998	3	0	-	-	-	-
	1999	4	0	-	-	-	-
‘I’iwi *Vestiaria obscurus*	1995	4	2	2	2	100.0	100.0
‘Oma’o *Myadestes obscurus*	1995	5	2	2	2	100.0	100.0
	1996	31	27	25	23	2.6	92.0
Hawai’I ‘Elepaio *Chesiempis sandwichensis*	1995	4	1	1	1	100.0	100.0
	1996	2	1	0	0	0	0
	1998	6	3	2	2	66.7	100.0
	1999	13	10	8	7	80.0	87.5
Palila *Loxoides bailleui*	1996	32	22	21	11	95.5	52.4
	1999	5	5	1	0	-	-
Puaiohi *Myadestes palmeri*	1996	7	5	5	5	100.0	100.0
	1997	25	10	10	10	100.0	100.0
	1998	38	26	26*	23*	92.3	88.5
	1999	18	8	5	5	62.5	100.0
‘Akohekohe *Palmeria dolei*	1997	6	6	6	5	100.0	83.0
Hawai’I creeper *Oreomystis mana*	1997	4	4	4	4	100.0	100.0
	1998	5	5	5	5	100.0	100.0
	1999	2	0	-	-	-	-
Maui parrotbill *Pseudonestor xanthophyrs*	1997	1	1	1	1	100.0	100.0
	1999	2	2	2	2	100.0	100.0
‘Apapane *Himatione sanguinea*	1997	7	2	2	2	100.0	100.0
‘Alala *Corvus hawaiiensis*	1993	11	8	7	7	87.5	100.0
	1994	6	6	6	5	100.0	83.3
	1995	1	0	0	0	-	-
	1996	29	12	7	6	58.3	85.7
	1997	29	15	10	9	66.7	90.0
	1998	16	10	4	4	40.0	100.0
	1999	15	5	2	2	40.0	100.0
Hawai’I ‘Akepa *Loxops coccineus*	1998	2	1	1	1	100.0	100.0
	1999	5	5	5	4	100.0	83.3
Nene *Nesochen sandvicensis*	1998	73	37	34	31	91.9	91.2
	1999	57	20	18	15	90.0	83.3

Note:

1 *** - includes nestlings collected from the wild and captive parent–reared chicks.**

2 **Viable eggs are fertile eggs containing normal, healthy embryos at the time of collection. Inviable eggs are infertile, cracked, have abnormal or broken aircells and membranes, or contain dead, dying or traumatized embryos.**

restoration efforts to ensure the long–term survival of released 'Alala in the future.

Hawaiian honeycreepers

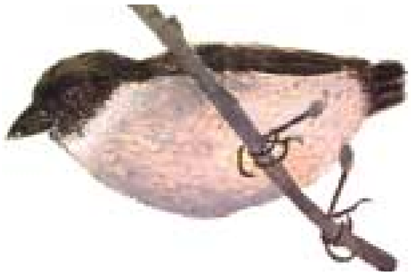
Perhaps the most spectacular of Hawai'i's endemic avifauna is the sub-family of honeycreepers (Drepanidinae). Reproductively isolated from the mainland populations and from each other on their respective Hawaiian islands, this group evolved into more than 50 unique species and subspecies. Many of these are now extinct, with the majority of the remaining taxa threatened with extinction. In order to test the effectiveness of captive–rearing and release strategies for this sub-family for future restoration efforts in Hawai'i, a pilot study was conducted with the Common 'Amakihi in forests where introduced avian disease and mammalian predators were present. Methodology used resulted in the first successful artificial incubation, hatching and rearing of a Drepanidinae. Sixteen chicks were hatched (mean hatch weight=1.4 g) and reared. Two different release strategies were evaluated for small honeycreepers; a 10–14 day acclimatization period in a hacking aviary ($4m^2$) in the native forest with subsequent food supplementation (soft release) and a two day adjustment period in small field cages ($1m^2$) with food supplementation. Although all the birds survived the initial release and returned for food supplementation, 12 of the 16 birds succumbed within 30 days to malaria infections, and four birds were not seen nor bodies recovered after 14 days. This is a clear demonstration that although propagation techniques can be successful, recovery will not succeed unless mosquito–free, predator-controlled re-introduction sites are available or strategies are developed to decrease mortality in naïve honeycreepers exposed to disease after release (Kuehler, *et al.*, 1996). However, the experience gained in the incubation and rearing of the Common 'Amakihi has subsequently provided the technology to hatch and rear six additional species of honeycreepers, the smallest being the Hawai'i 'Akepa with an adult weight of 9–11g. and a hatch weight of 1.0g.

Hawaiian thrushes

Very similar to the mainland solitaires, five species of Hawaiian *Myadestes* thrushes survived until very recently. However, it is now thought that only two of these species persist; the 'Oma'o on the Big Island of Hawai'i and the Puaiohi on the island of Kauai. In 1995 and 1996, the first restoration attempt of a small Hawaiian passerine in disease-free, predator controlled habitat was made with the release of captive–reared \`Oma\`o, into the Pu\`u Wa\`awa\`a Forest Reserve; habitat that has been without this species for nearly 100 years (Fancy *et al.*, in press). In 1995, two birds were re-introduced as a preliminary test release and in 1996, 23 birds were released in cohorts numbering from 2–7 birds. Of the 25 released birds, 23 are known to have survived 30 days (life of the transmitters). Follow-up surveys in 1997 and 1998 indicate that many released \`Oma\`o have survived to sexual maturity and have bred.

The Puaiohi is an endangered thrush, endemic to the island of Kauai and restricted to the Alaka'i Wilderness Area above elevations of 914 m. a.s.l.. Since 1995, this Hawaiian solitaire has been the focus of an aggressive recovery effort that has incorporated the funding, field efforts and the captive propagation and release expertise of several governmental and private agencies.

In 1996 and 1997, 15 Puaiohi eggs were collected from the wild, hatched and reared at the Keauhou Bird Conservation Center; becoming a captive breeding flock in 1998 and 1999, producing 28 chicks. In early 1999 and again in early 2000, a total of 19 of captive–reared chicks were transported to Kauai, acclimatized for 14 days in hacking aviaries ($3m^2$), transmittered and soft–released from two release sites in the Alaka'i Wilderness Area. Although supplemental food was offered at the hacking cages, only a few of the birds returned to feed. All 19 birds survived to independence and survived for at least 60 days. At least eight of the 14 birds in the 1999 release flock formed six pairs, including pairs made of captive–captive birds, captive–wild birds, and a trio of captive birds (1:2). From these pairings, 21 nesting attempts were made, successfully fledging seven chicks. This is the first release program for a passerine that has successfully incorporated all of the following techniques to include: collection of wild eggs, artificial incubation and hand–rearing, captive–breeding, release and subsequent breeding of the released birds in native habitat. This complete re-introduction scenario for the Puaiohi; from the wild to captivity and back to the wild, where breeding has been confirmed on several occasions, has occurred over only three years time, a remarkably successful recovery action (Kuehler, *et al.*, in review).



Po'ouli © Tonnie Casey

The first seven years of this program presents a more optimistic future for the beleaguered avifauna of the Hawaiian islands. As the captive flocks of the endangered species grow, and the techniques for rearing and release are refined, it is hoped that many of the endangered Hawaiian birds will benefit from restoration efforts. However, it must be emphasized that captive propagation and re-introduction is only one aspect of the ecosystem management tools required in Hawai\`i to conserve and restore endangered native bird species.

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Experimental releases of Hispaniolan parrots in the Dominican Republic: implications for Puerto Rican parrot recovery

Thomas White et al.

Introduction
Formerly abundant throughout Puerto Rico, the Puerto Rican parrot *Amazona vittata* is now considered one of the 10 most endangered birds in the world. Currently, there exists only one wild population of approximately 35–40 individuals in the Caribbean National Forest of eastern Puerto Rico. Two additional captive populations totaling around 105 birds are currently held in separate aviaries located in the Caribbean National Forest and the Rio Abajo Commonwealth Forest in north central Puerto Rico. The primary function of these captive populations is to provide a sustainable source of Puerto Rican parrots for release into the wild to bolster the current wild population, as well as for eventual re-establishment of a second wild population elsewhere in Puerto Rico (Snyder *et al.*, 1987). Releases of captive–reared Puerto Rican parrots to augment the wild population have long been recognized and recommended as a crucial step toward recovery of the species (U.S. Fish and Wildlife Service, 1982).

Captive-reared Puerto Rican parrots were previously released in the Caribbean National Forest in 1985. However, the small number (three individuals) released was insufficient to evaluate viability of the technique to achieve Puerto Rican parrot recovery goals. A similar release of 18 captive–reared Hispaniolan parrots *Amazona ventralis* was conducted in the Dominican Republic in 1982, but also was inconclusive because of logistical problems, short study duration (two months) and unstructured pre–release protocols (Snyder *et al.*, 1987 & Collazo *et al.*, 2000).

From 1997 to 1999, we released 49 captive–reared Hispaniolan parrots in Parque Nacional del Este, a 42,000 ha area of subtropical dry and moist forests located in southeastern Dominican Republic (Abreu & Guerrero, 1995). Each parrot was radio–transmittered and monitored for up to one year to determine survival, movements and habitat use. Our goal was to develop a release strategy for Puerto Rican parrots and gain insights about potential survival of released captive–reared Puerto Rican parrots. We used Hispaniolan parrots because they are the closest relatives of Puerto Rican parrots, are not critically endangered, and have been used successfully as surrogate parents for Puerto Rican parrots (Snyder *et al.,* 1987). Moreover, their use for experimental releases in the 1980's supports our contention that hispaniolan parrots are a suitable biological model from which to gain insights about Puerto Rican parrot post–release survival. The value of the hispaniolan parrots as a Puerto Rican parrot model was enhanced in this case because released parrots were reared in the same aviaries as Puerto Rican parrots destined for future releases in Puerto Rico. We believe that the intrinsic demographic and genetic value of captive Puerto Rican parrots for the recovery of the species precluded their use during the developmental phase of a release strategy. Finally, because Hispaniolan parrots are native to the Dominican Republic, the releases were conducted there in order to release parrots in historical occupied habitat, and to avoid exacerbating the problem of introduced exotic psittacines in Puerto Rico.

Here we present a general overview of the release project, including techniques and ideas that worked, as well as those which did not. Readers are referred to Collazo *et al.* (2000) for detailed descriptions of study area, experimental design and results of statistical analyses. As with most field projects, many problems we encountered were initially unforeseen, and some early ideas later proved impractical.

Problems were encountered during the initial planning phase, during which we intended to conduct multiple releases in four widely separate areas of the Park. Although this approach would have provided for spatial and temporal replication, practical limits of both personnel and equipment later rendered this impossible. Instead, we chose to focus efforts within an area of 5,000 ha encompassing the northwestern quadrant of the Park. However, by foregoing true spatial replication we later were able to more intensively monitor each released parrot, thereby gaining detailed data on individual survival, movements and behavioral interactions.

Release techniques
We released parrots from four separate release cages which also were used as on-site training and acclimation facilities. Measuring 3.6m long x 1.5m wide x 2.1m tall, each cage contained four parrots and provided space for flight. Cages were suspended approximately 2m above ground level. Parrots were acclimated on-site for a minimum of 40 days, during which they were exposed to a wide variety of locally occurring native foods. Use of cultivated agricultural products was avoided, as the objective was to accustom parrots only to those species they would later encounter within the study area and to minimize the possibility that they would become local crop depredators. Each parrot also was equipped with a "dummy" radio-collar of the same weight (11 g) and configuration as the actual radio-transmitter in order to accustom them to the device before release. Parrots also were subjected to an exercise program (e.g., forced flight) during the acclimation period in an effort to maintain or increase flight stamina and ability. Approximately 2–3 days prior to release, each parrot was subjected to a complete veterinary examination and functioning radiotransmitters were attached. On dates of releases, cages were opened before dawn and parrots allowed to exit at will.

Results of releases
Of the 24 parrots released during 1997, five died within five days of release (Collazo *et al.*, 2000). Five additional parrots died shortly after onset of the marked dry season (January–April) characteristic of eastern Caribbean dry forests. Two additional birds fell prey to a Red-tailed hawk(s) *Buteo jamaicensis*. In contrast, none of the 25 parrots released in 1998 died within five days of release. In fact, birds of the first 1998 release (29th June 1998) had already survived 10 weeks when Hurricane Georges hit Parque del Este on 22nd September 1998. Effects of this hurricane on subsequent parrot survival are detailed in Collazo *et al.*, (2000).

We report 2 modifications to pre–release training and conditioning protocols that may have contributed to inter-annual differences in early survival trajectories. During the 1997 releases, we felt that the parrots did not exhibit good flying skills. Thus, we subjected 1998 birds to a more rigorous exercise routine. Median keel scores (index of flight muscles) increased significantly (P=0.0002) (Collazo *et al.*, 2000) from 3.0 in 1997 to 3.5 in 1998. The second pre-release modification consisted of reducing blood samples collected 2-3 days prior to release in 1998 (i.e., 1 vs. 2 cc per bird in 1997) or not collecting a sample at all (random selection). Although parrots can replace 2 cc of blood within 3–7 days, it is possible that birds released in 1997 were weaker when released than birds in 1998.

Our work sheds light on the importance of timing of release. We found that survival rates measured over the dry season were higher for birds released in October than in December. A plausible explanation for these differences may be that birds released in October had a longer opportunity to exploit higher levels of food availability. Eastern Caribbean phenological data suggest that food availability in moist and dry forests is greater during late summer–fall (rainy season) than during winter–early spring (dry season) (Lugo & Frangi, 1993). Conversely, factors such as presence/absence of predators were deemed similar for both groups. Although further tests are necessary before definitive inferences are drawn from this "seasonal food hypothesis", prudence dictates that releases take place within the widest possible food availability window.

Conclusion

These results have been incorporated into the pre-release training and acclimation of Puerto Rican parrots scheduled for release during the summer of 2000. For example, on–site acclimation cages in Puerto Rico have an internal volume twice that of cages used in the Dominican Republic. This allows additional flight space per bird and facilitates maintenance of flight ability and stamina prior to release. Birds will be subjected to forced flight training at least as often and intensive as during 1998 pre-release training in the Dominican Republic. Pre-release physical exams will be conducted 5–7 days prior to release (as opposed to 2–3 days) and blood samples collected will be limited to 1.0 cc per bird to avoid potentially weakening birds immediately prior to release. Finally, predator aversion training will be conducted using a live red–tailed hawk while birds are housed at the actual release site. We hope that these modifications and measures will further aid in reducing or eliminating early, post–release mortalities as demonstrated by the successful 1998 releases in the Dominican Republic.

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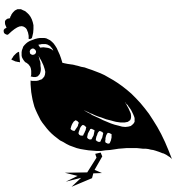


Re-introduction of peregrines in the eastern United States: an evaluation

Tom Cade, Harrison Tordoff & John Barclay

Introduction and background.

The recovery of the peregrine falcon *Falco peregrinus* in North America to the point at which it could be de-listed as an endangered species in 1999 was one of the great conservation achievements of the 20th Century. It was accomplished by the cooperative efforts of dozens of non-governmental organizations, state, provincial, and federal agencies, and many hundreds of private citizens in Canada and the United States. In this paper we focus on the re-introduction of peregrines into the eastern United States, one part of the continental program to restore this species to its rightful place in nature.

Although always rare, the peregrine was rather widely distributed through the eastern third of the United States, east of 90° W long., prior to the widespread use of organochlorine pesticides beginning in the late 1940s. Peregrines bred in fairly substantial numbers from Maine south through the mountains of New England, the Adirondacks of New York, and the Appalachian system to northern Georgia and Alabama, as well as along major rivers, wherever suitable nesting cliffs occurred. They also nested on cliffs in the western Great Lakes region, as well as on river bluffs of the upper Mississippi and its tributaries. A scattered population nesting in tree cavities once occurred along the Ohio River and lower Mississippi River valleys down into the cypress swamps of western Tennessee and northern Louisiana, but these birds had largely disappeared by 1900 with the loss of the big riparian trees.

Hickey (1969) described a total of 275 known nesting locations in the USA east of the Rocky Mountains, and Berger *et al.* (in Hickey 1969) listed 205 sites in the greater Appalachian region, where Hickey thought there might actually be 350 or more falcon territories. In the upper Mississippi River valley and western Great Lakes region there were some 40 known eyries and an estimated population of c. 50 nesting pairs (Redig and Tordoff in Cade *et al.*, 1988). Thus, the total number of available nesting territories in the eastern third of the United States probably ranged between 400 and 450 locations, approximately 80-90% of which were occupied in any given year in a region of >3 million km^2.

In the late 1940s and early 1950s local field observers began to notice a decrease in the number of occupied falcon territories, and by 1964 not one nesting pair could be found in the entire region (Hickey, 1969). At the same time the species had become greatly reduced in number in the western United States. These unprecedented losses paralleled similar declines in peregrine numbers in Europe and soon came to be associated with both lethal and sublethal effects of organochlorine pesticides, particularly DDT (Hickey, 1969 & Cade *et al.*, 1988). Consequently, the U. S. Fish and Wildlife Service listed both the *tundrius* and *anatum* subspecies as "endangered" in 1970. The Service then developed four regional recovery plans for restoration of the peregrine falcon. The beginning objective of the Eastern Peregrine Falcon Recovery Plan was to re-establish a population equal to half of the estimated original population of 350 nesting pairs or to whatever number the current environment would support. It was later modified to specify 175 to 200 nesting pairs with a minimum of 20-25 pairs in each of five recovery areas, demonstrating successful, sustained reproduction for a minimum of three years.

Recovery strategy.
The eastern recovery plan was unique in that it had to rely on the re-introduction (introduction) of peregrines from non-indigenous sources, because the native falcons (formerly referred to as "duck hawks") had been extirpated before recovery efforts began. The strategy recommended was to maximize the genetic diversity of a captive stock by interbreeding individuals from various geographic and subspecific sources, so that the released progeny would provide natural selection with a wide array of phenotypes (and genotypes) to act upon, thus creating a new, viable population adapted to the current conditions of the eastern environment.

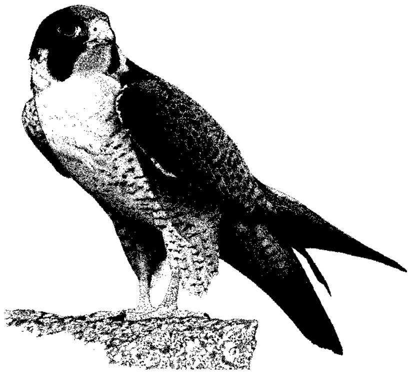
The U.S. Fish and Wildlife Service adopted an official policy, specific to the eastern recovery region, that allowed for the propagation and release of any peregrine falcon of North American origin (*pealei*, *anatum* and *tundrius*) and further provided for the support and funding of the same activities involving falcons "from other geographic regions" (i.e, exotics) for specific research purposes on a case by case basis.

Most of the birds subsequently released were either of *anatum* or *tundrius* ancestry, or mixtures of the two. A fair number of Peale's falcons were also used in various combinations, followed by lesser numbers of foreign birds- *brookei*, *cassini*, *peregrinus*, and *macropus*, also mostly interbred with American birds (Temple in Cade *et al.*, 1988 & Tordoff and Redig, in press).

Results and discussion.
The captive breeding and releases were carried out by two private sector programs. The one at Cornell University's Laboratory of Ornithology, operated by The Peregrine Fund, Inc., involved maintenance of a large breeding collection of captive falcons and the release of their progeny into >15 eastern states and the District of Columbia. Some 1300 young falcons were released by hacking and fostering between 1974 and 1998. The program at the Raptor Center, University of Minnesota, relied on offspring produced by 35 private falcon breeders and involved the release of more than 900 falcons in 12 Midwestern states from 1982 through 1999. We estimate that c. 80% of the released falcons survived to independence and around 40% survived their first year in the wild (Grier and Barclay in Cade *et al.*, 1988 & Tordoff and Redig, 1997).

Our first strategy was to release and re-establish peregrines at their natural, cliff eyries along rivers and in mountains, but depredation by the great horned owl *Bubo virginianus* caused high loss of fledglings and first-year birds and thwarted the attempts of several pairs to establish eyries on cliffs, especially along lowland rivers. By locating hack sites at higher elevations in the northeastern mountains, at owl-poor sites along Lake Superior, in urban/industrial areas, and in coastal salt marshes we greatly reduced contact with owls, and the released peregrines survived much better.

Returning subadult and adult peregrines often established territories at hack sites and acted aggressively toward released young, necessitating the development of a new hacking location. We learned that this problem could be reduced by releasing young at sites where peregrines were unlikely to establish territories but which were adjacent to several suitable nesting sites.



Peregrine Falcon *Falco peregrinus*
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Another potential problem was the persistence of organochlorine residues in the environment and their continued occurrence in the falcon's prey. Even though the governments of Canada and the United States rescinded the use of DDT and dieldrin between 1969 and 1974, residues of these compounds have remained in the environment to the present, and in some areas peregrines still show the influence of DDT on eggshell thinning, and the reproduction of some pairs may still be reduced as a consequence. Overall, however, these impacts have not influenced reproduction enough to prevent vigorous growth of peregrine populations in North America since the late 1970s.

The habitats that peregrines use in the eastern USA have changed dramatically since the 1800s. In particular, the eastern forests have recovered much of the open, agricultural lands that existed 100 years ago, and many of the peregrine's historically occupied cliffs are now heavily timbered. One wonders how these changes may have influenced the present day acceptability to peregrines of some of these old territories that remain unoccupied. It seems likely that forest regeneration has favored great horned owls at the expense of the falcons. In hindsight, increased owl depredation may have been a factor responsible for the reported abandonment of some falcon eyries in the 1930s and 1940s prior to use of DDT. Other cliffs have been overtaken or marginalized by human actions.

The first released peregrines in the East bred successfully in the wild in 1980, and in 1987 in the upper Mississippi region. Since then these re-established breeding populations have grown rapidly, so that by 1999 there were 156 territorial pairs in the eastern USA and 92 pairs in the Midwestern states, a total of 248 pairs east of 90^{o} W long. These populations have continued to increase at annual rates of c. 10% even though most releases occurred prior to 1992 in the East, and there is every reason to believe that they will continue to grow for some years to come. The final, regional population at carrying capacity will surely exceed the historically known numbers; indeed, they have already been exceeded in the Midwest.

The breeding distribution of the newly established peregrines is somewhat different from the original distribution of the extirpated duck hawks. This difference results in part from the failure, so far, of the re-established falcons to reoccupy successfully eyries on the lowland river systems and in much of the Appalachian Mountains, and in part to their occupation of previously unused nesting habitats in urban/industrial areas and in the salt marshes of the Atlantic Coast. Nesting falcons now occur on 56 cliffs in the northeastern states, on 13 cliffs in the southern Appalachians, and on 10 cliffs in the western Great Lakes region of the USA. Approximately 28 pairs breed on former hack towers in the salt marshes of New Jersey, Maryland, and Virginia. The remaining 141 pairs nest on buildings, bridges, and power plant smokestacks in urban/industrial areas spread throughout the eastern and midwestern states.

Although these re-established peregrines come from parents with diverse genetic and geographic origins external to the eastern United States, they have converged rather closely to the former, indigenous duck hawks in their habits and ecology, except that some birds of northern, migratory stock winter farther south than the duck hawks did. Survival, productivity, natal dispersal, exchange of individuals among remote populations, and persistence of nesting territories over many years are all characteristic of a viable, regional peregrine population (Tordoff and Redig, 1997 & Corser *et al.*, 1999).

Conclusion

Successes and failures: This newly established falcon population shows all the signs of viability required for continued existence into the Indefinite future. Its size has now exceeded the stated recovery goal and approaches that of the original duck hawks; it continues to increase at a rate of c. >10% per year with little or no further augmentation. The eastern and midwestern populations exchange individuals (and genes) between themselves, as well as with Canadian populations to the north.

The main shortcomings of the program relate to distribution rather than to population size and demographic viability. So far the re-established falcons have been unable to occupy the former breeding locations along the lowland river systems and in the central Appalachians owing to predation by abundant great horned owls. This same sort of exclusion has been noted in Europe where eagle owls *Bubo bubo* occupy habitats suitable for nesting peregrines. Also, our tactical assumption that falcons released from towers in the coastal salt marshes, and the progeny of established pairs there, would disperse to settle on inland cliffs has proved to be incorrect, as these birds have shown little inclination to move away from the coast, or else those that do disperse inland succumb to owls.

Conservation perspective: Species restoration, like habitat restoration, is usually a compromise between the ideal of return to the original, pristine condition and the practical limitations of what is possible in a drastically altered, human-dominated landscape. The habitats available to peregrines in the eastern USA today are not entirely the same as the habitats in which the original duck hawks settled and evolved. While the natural cliffs that served as eyries before World War II have not been re-occupied in many parts of the range, the re-established falcons have found other areas in coastal salt marshes, in cities, and in industrial zones where they can survive and reproduce. The overall eastern population is certainly capable of existing into the indefinite future, and as natural selection continues its work, eventually a peregrine that can coexist with owls may yet emerge to regain the cliffs along the lowland rivers and in the Appalachian Mountains. Until then, the peregrine will be there in the eastern environment to occupy its unique niche in the community of living organisms and to be seen and admired by *Homo sapiens*, the wise species that refused to allow this fellow globetrotter to pass into oblivion.

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Captive propagation and *in-situ* crane conservation in the USA and Russia

George Archibald

Introduction

Prolific captive populations of the endangered Mississippi Sandhill crane *Grus canadensis pulla*, whooping crane *Grus americana*, Siberian crane *Grus leucogeranus*, red-crowned crane *Grus japonensis*, and white-naped crane *Grus vipio* have been established in the USA (all five species) and in Russia (Siberian, red-crowned and white-naped cranes).

Using captive-produced stock efforts have been made to start new wild populations of whooping cranes in western USA and in Florida, and to bolster existing populations of the remaining four species. The first experimental population of Whooping cranes was established by substituting both captive-produced and wild-produced eggs into the nests of greater sandhill cranes *Grus canadensis tabida* at Grays Lake National Wildlife Refuge in Idaho. Although 77 Whooping cranes were reared to migrate with their foster parents, conspecific pairing did not occur. It is believed that sexual imprinting on foster parents prevented conspecific pairing. However, Whooping cranes learned to thrive in the more upland foraging niche of the sandhills and they learned a new migratory route from the foster-parents. The experiment was discontinued and only a few birds survive in the population.

Starting in 1993, efforts have been underway in Florida to start a second and non-migratory population of wild whooping cranes. Captive produced cranes are either parent-reared or crane "costume-reared" and then "soft-released" from holding pens in Florida. Parent-reared cranes were reared by captive pairs of whooping cranes. "Costume-reared " birds were reared by keepers cloaked in white cloth and with one hand supporting a puppet that resembled the head of a crane. "Soft released" birds are flight-restricted and then held in a large enclosure in a wetland in the release area. After becoming accustomed to their new surroundings, wing brails are removed and the cranes gradually fly from the release enclosure.

Approximately 170 whooping cranes have been released in Florida from which about 64 have survived in January of 2000. Predation from bobcats is the single greatest source of mortality. In 1999, two pairs laid eggs but these attempts to breed failed through flooding and predation on eggs. Eleven pairs have formed at the time of this writing.

In western Siberia since 1991, efforts gave been made to bolster the dwindling numbers of Siberian cranes through the releases of captive-produced cranes on the breeding grounds, migration resting areas, and on the wintering grounds. Siberian crane eggs were substituted into the nests of Eurasian cranes with the hope of producing guides birds to lead costume-reared Siberian cranes along new migration routes to new wintering areas. Captive-produced costume-reared and parent-reared juveniles were released with wild Siberian cranes on the breeding grounds and migration staging areas. They joined both wild Eurasian cranes and wild Siberian cranes and migrated. Both costume-reared and parent-reared juveniles were released with the wild cranes on wintering grounds in Iran and India. In subsequent years there has been only a single unconfirmed sighting of a bird released on the breeding grounds. None of the birds released on the wintering grounds migrated in spring.

One-year old hand-reared red-crowned cranes and white-naped cranes, were released with wild cranes at Khiganski Nature Reserve, Russia. Some cranes joined the wild cranes and migrated south with them in autumn. Subsequently several of these cranes were observed on the breeding grounds and pair formation with wild cranes has occurred.

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WEST ASIA

The re-introduction of houbara bustards in the Kingdom of Saudi Arabia

Philip Seddon et al.

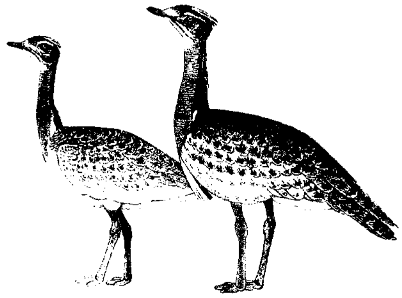
Introduction

Some of the earliest references to falconry in the Middle East date back more than 1,000 years, preserved in the sayings of the Prophet Mohammed, and the flying of trained falcons has been pursued with a passion throughout the Arabian Peninsula right up to modern times. Although there has been a change in emphasis from pragmatic falconry to obtain food, to a leisure sport, one constant has been the favoured quarry—the houbara bustard *Chlamydotis* sp.. Cryptic and unspectacular, the houbara bustard is a medium-sized gruiform inhabiting the deserts and semi-deserts of North Africa and west and central Asia. Current thinking divides the houbara complex into two species and a subspecies; the form found in Asia is the species *Chlamydotis [undulata] macqueenii*.

Conservation status of houbara bustards

Although accurate assessment of houbara population sizes is hampered by both the behaviour and the habitat of the species, there are indications that tens of thousands of houbara occupy breeding grounds throughout west and central Asia—sufficient numbers for the houbara to have been removed from any category of threat in the 1998 IUCN Red List. However, the recent spread of large–scale falconry expeditions into previously isolated regions in the Commonwealth of Independent States, along with trapping of live birds for the training of falcons in the Middle East have lead to actual and suspected population declines throughout the species range, and have prompted a re-assessment of global threat status.

Within the Kingdom of Saudi Arabia the houbara bustard was once considered to be a widespread breeding resident, distributed over large areas of the north and central desert plains. Unregulated hunting, with both falcon and firearm, together with habitat loss to agriculture and urban development, meant that by the latter half of the 20th century houbara had been virtually eliminated as a resident species, remaining only as one small breeding population in the basalt steppe of the far north (Seddon *et al.*, 1995). Migrant houbara, presumably from Central Asia, still enter the Kingdom during the winter months and support the reduced recreational falconry that still takes place.



Houbara bustard
Chlamydotis [undulata] macqueenii
© P. Paillat

Houbara bustard restoration in Saudi Arabia

It was as a symbol of a traditional way of life that the houbara bustard was chosen to be the focus for one of the Kingdom's first intensive species restoration projects. In 1986 the National Commission for Wildlife Conservation and Development (NCWCD) was established to oversee the creation and management of a network of protected areas, and to undertake the restoration of native species within such sites. The virtual absence of breeding populations of houbara in Saudi Arabia meant that the restoration of the species could realistically be achieved only through a program of re-introduction, involving the release of captive–bred birds.

The task of breeding and releasing houbara was handed to the National Wildlife Research Centre (NWRC), one of the research stations operating under the auspices of the NCWCD. Fertile eggs for the captive breeding program were collected under Government permit from resident populations in the Baluchistan region of Pakistan during 1986 to 1988. By 1991, through the application of artificial insemination techniques the NWRC was able to produce enough houbara chicks to replace losses in the breeding unit and to begin trial releases (Saint Jalme & van Heezik, 1996).

Concurrent efforts had meanwhile been taking place to select and prepare a suitable release site. In 1988 the Mahazat as-Sayd protected area was established, and in 1989 the entire 2,200+ km^2 area was enclosed within a fence. Although previously overgrazed by domestic livestock the vegetation within Mahazat as-Sayd recovered rapidly and by 1991 was supporting trial re-introduction programs for sand gazelle *Gazella subgutturosa* and Arabian oryx *Oryx leucoryx*, as well as houbara.

The re-introduction process: pre-release preparations and post–release survival

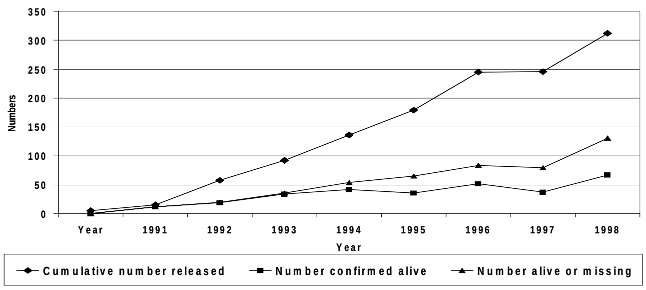
Re-introduction trials between 1991 and 1994 tested four techniques: release of adults; release of broods; release of feather–cut sub–adults (<1 year old), and release of flying sub-adults. With the exception of a small-scale hard release of adults in 1991, all releases took place within a 400 ha predator-proof enclosure within the reserve from which houbara were free to fly. It soon became evident that predation was the major problem. Avian predators killed almost 2/3 of the feather-cut sub–adults when still inside the pre–release enclosure, while broods were vulnerable to avian predators within the enclosure, and avian and mammalian predators outside. The greatest success rate (48%), measured as persistence of the birds within the reserve, was achieved with the release of flying sub-adult birds, and by the end of 1994 Mahazat as-Sayd had a free-ranging population of 35 houbara (Combreau & Smith, 1998).

An interesting observation to arise from these early studies was the fact that houbara, once they had been flying within the reserve for some time, no longer fell prey to mammalian predators. The amount of time after which they appeared safe from predation was dependent on the density of foxes *Vulpes* sp. in the vicinity of the pre–release enclosure; without predator control within two weeks of release those houbara that were going to fall prey had done so, but with predator control this period of vulnerability was as long as six months. This suggested that something had taken place during those first weeks or months to make individuals apparently less susceptible to predation; possibilities included better use of habitat, or greater awareness of potential predators, perhaps through observation or escape. In addition, early deaths to predation would have included a proportion of poorly adapted individuals—a weeding out of unfit birds that would not ordinarily take place in the pre-release period of captive care.

Work continued during 1995 to 1997, releasing sub–adult houbara into the pre–release enclosure after a period of 1–4 months in small soft pens at the release site. A further problem that this method entailed was one of timing: houbara hatching within the NWRC breeding unit peaks in April, therefore sub–adult birds of 2–4 months would be ready for release between June and August, the northern summer, when ambient temperatures in Mahazat as-Sayd may reach 46°C. Houbara were therefore being released at the hottest, driest time of the year, an added stress that may have meant that some of the recorded predation was actually the opportunistic taking of weakened birds. A number of programs attempted to reduce the severity of this stress, including a gradual transition from pellets to a more natural diet; reduction in availability of water during the pre–release holding period (free–ranging houbara do not drink), and the use of temporary release sites situated in relatively green parts of the reserve. Despite this however, annual post–release mortality reached 50% or more, with the principal predator believed to be the red fox *Vulpes vulpes*. Since 1993 potential predators of houbara had been systematically removed from a zone around the pre–release site- the aim being to remove individual predators that may have become keyed in on houbara releases. A reserve-wide program of predator eradication was not undertaken for two reasons: (1) the reserve was established to protect also a number of native mammalian carnivores, including sand cat *Felis margarita* and Ruppell's fox *Vulpes ruppelli*, and (2) for the wider aims of the houbara re-introduction project to be achieved, i.e. the restoration of resident breeding population of houbara throughout suitable habitat in Saudi Arabia, released houbara would ultimately have to survive, as do wild birds, alongside red foxes and other predators.

Although by 1995 the reserve contained a resident houbara population of more than 50 birds (Fig. 1), it was felt that post-release survival could be improved through more active means. Between 1995 and 1998 a sample of birds released were subject to pre-release predator awareness training. In 1995 this took the form of simulated predator attacks using a taxidermic model of a red fox and taped houbara alarm calls. Apparent habituation to model attacks and no evidence of improved post-release survival in the trained versus a control group lead in 1996 to the use of a restrained live red fox. Significantly more houbara in the trained group survived during the 3-6 months post-release than did untrained birds, suggesting that early experience of an encounter with a predator was an important factor in individual survival (van Heezik *et al.*, 1999).

Figure 1. Re-introduction of houbara bustards into the Mahazat as-Sayd protected area, Saudi Arabia. Note that "number confirmed alive" is derived from telemetric monitoring; "number alive or missing" includes birds whose radio-transmitters are believed to have failed; therefore these two parameters form lower and upper limits to estimated actual houbara bustard population size within the protected area (excluding recruitment).



During 1997 and 1998 an outbreak of rabies was recorded in the Mahazat as-Sayd fox populations, affecting mainly the more gregarious red fox and resulting in measurable decreases in red fox numbers. Possibly as a consequence, post–release survival of houbara was high (72% after one year) in the 1998/99 release season, and there was no difference between predator trained and untrained birds. However, several features of the release programme differed at this time. Instead of small pre–release pens, large flight cages were used in the pre–release area. This allowed the birds to be held for longer and afforded them opportunities to exercise flight muscles. The pre–release period was extended to avoid releasing birds in hot, dry conditions, allowing researchers to wait until the first autumn rains. Unfortunately the rains expected in November 1998 did not arrive and the birds were released only early in 1999, at almost one year of age. So higher survival in 1999 could have been due to the greater age or fitness of the birds; the relatively cool, wet conditions, and/or lowered predator densities. In order to start to pick these various factors apart, in 2000 the release conditions have been maintained, i.e. birds hatched in 1999 have been held in large cages awaiting spring rainfall. No predator training has taken place, and recorded fox densities indicate population recovery. As at June 2000 release of 99 birds indicates only slightly increased post-release mortality due to predation compared with 1999.

The importance of post–release monitoring

A key factor enabling NWRC staff to assess the success or failure of the different phases of the houbara release program has been a policy of radio–tagging all released birds. Birds are fitted with backpack-mounted solar-powered units, with a maximum recorded life of over six years. This has been a major, but necessary expense, since once released the houbara become virtually invisible. Today you could drive through Mahazat as-Sayd for days and never encounter nor flush a single houbara; you would conclude that the reserve had no houbara, whereas radio-tracking reveals a resident population of over 100 birds (Fig. 1). Radio-tagging has been essential, not only to assess post–release survival, but also to follow the birds once they become sexually mature at 1 to 2 years of age. In 1995 the first houbara nest was located in Mahazat as-Sayd by tracking an adult female (Gelinaud *et al.*, 1997). Since that time the re-established houbara population has bred each year, and research attention has turned to assessment of chick survival and recruitment into the breeding population. Preliminary population viability modelling indicates that with maintained high adult survival and only modest breeding success, the Mahazat as-Sayd houbara population has a high probability of growth and persistence.

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OCEANIA

Re-introduction of the Campbell Island teal, New Zealand

Peter McClelland

Introduction

Although first described in 1886, only two Campbell Island teal *Anas nesiotis* were ever seen prior to their discovery on Dent Island, a 26 ha tussock covered island 3 km off the main island, in 1975. Campbell Island teal are a small flightless brown duck weighing around 300g for females and 450g for males. While originally regarded as a subspecies of *Anas aucklandica* (as were Brown Teal and Auckland Island Teal), genetic work (Daugherty *et al.*, 1999) indicates that Campbell Island Teal and the other teals all warrant full species status (Auckland Island teal as *Anas aucklandica* & New Zealand brown teal as *Anas chlorotis*) and this is how they are managed.

Current estimates put the wild population at probably less than 25 pairs, and if the latest search carried out in 1998 is anything to go by, it could easily be much less. Even in the best scenario the population is unlikely to be more than 100 individuals as this is the estimated maximum holding capacity of the island if all available habitat was used. This low number and the risk of the population being hit by a stochastic event, such as the land slips which had occurred in 1976 destroying a significant area of habitat, prompted a plan to establish a captive population as a backup.

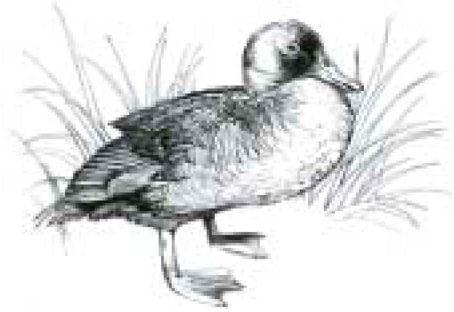
In 1984, three males and one female were caught and returned to New Zealand. However although a range of pairings were tried, no eggs were produced. This in turn prompted the writing of the Subantarctic Teal Recovery Plan (McClelland, 1989), which covered both Campbell Island teals and their less endangered cousin the Auckland Island teal. Eight Auckland Island teal had also been bought back in 1984 to act as analogues for developing breeding techniques. Some of the Auckland Island Teal bred and the captive population is now being phased out as it has filled it's role and the species is not at risk in the wild.

The Recovery plan set as it's main objective the returning of Campbell Island Teal back to 12,000 ha Campbell Island. At the time this was thought to be unlikely for at least 10 years due to the presence of rats *Rattus norvegicus* and cats *Felis catus*, and our inability to clear an island of that size. This left three options:

- do nothing and either allow the birds to find there own way across to Campbell and re-establish, or to do a direct transfer at that time. These were not deemed acceptable options due to the likely length of time involved and the risk that a stochastic event could wipe out the species. Even when Campbell Island was available, there was no guarantee that enough birds could be sourced directly from Dent Island for a transfer to succeed.
- direct transfer to another island to establish a backup population. As with a) there was no guarantee that sufficient birds could be sourced for the release without putting the Dent Island population at risk.
- continue with trying to establish a captive population by removing as few additional birds as possible while giving an acceptable chance of achieving your captive breeding goals. To this end it was set that up to five males and five females could be removed. There was the additional option of eradicating rats and cats and hoping that at some stage in the future teal would make their own way across to the main island. It was considered that this would take too long, if it happened at all.

There has been a reasonably long history of captive breeding of the brown teal for release in New Zealand, with over 1,200 birds being produced and released between 1985 and 1998. While the releases have not established any new populations, due to predation, the success of the breeding program itself showed the potential for Campbell Island teal. Much of the Brown teal work was carried out by private breeders and one of the most successful of these was selected to try breeding the first three birds after initial attempts at a departmental facility had failed. It is apparent that the failure of the birds to breed at either location was due more to the unwillingness of the female than any failings on behalf of the facilities.

In 1990 another expedition visited Dent and, with the advantage of taped calls, was able to return with three female and four male teal. Once again a variety of pairings was tried, including "training" the males with captive raised Auckland Island females. Although fertile hybrid eggs were produced, no Campbell females produced ducklings until 1995 when, using a multi–pair pen previously trialed with mixed-species pairs, one pairing bred. Since then a variety of pairings have produced young which incorporated genes from all of the males but still from only one of the original females. Mini satellite DNA profiling was used to access the genetic relationships of all wild birds in captivity. Nine birds shared 86% of the bands appearing in their fingerprints, showing, as one would expect in a population of this kind, a high level of background relatedness, and suggesting that the Dent Island population was itself sourced from a very small number of founders.



Campbell Island Teal *Anas nesiotis*

Once the captive population was established, providing set genetic and quarantine issues are dealt with appropriately, we were in a position to produce sufficient stock to carry out multiple releases either into the same site or different sites without affecting the captive breeding potential. The biggest risk of this was that Campbell Island would not be available for some time and that we would have, in effect, a aviary population with a corresponding decrease in their ability to survive once released to the wild. As such it was decided, in the recovery plan, that a holding island was required to enable captive-bred birds to adapt to living in the wild and hopefully to breed and provide more birds for the re-introduction to Campbell. Criteria were set to locate a holding island with habitat as close to Campbell as possible, this would ideally have been a tussock covered subantarctic island. However the ecological values of all the most likely islands excluded them i.e. pristine or near pristine. Due as much to availability as suitability Whenua Hou Nature Reserve (Codfish Island) a 1300 ha island 5 km off Stewart Island, was chosen. While being largely covered in tall forest, and lacking the more extreme climate and voracious aerial predators (skua) of Campbell, Codfish did provide a diverse range of freshwater and coastal habitats and was considered a suitable halfway house for adapting the birds to the wild.

In March 1999, 12 teal (4:8) (the sex ratio being based on availability of suitable birds) none of which had bred before, were released onto the island. All the birds had radio transmitters and were monitored sporadically for the year following release. However problems with batteries dying prematurely and aerials breaking meant that some birds could not be monitored as well as others. The birds were equally divided between a stream flowing onto a rocky coast and another flowing into a sandy bay. Geography meant that the birds on the rocky coastline were far harder to locate and even when located they were very difficult to follow to record behaviour. Of the five females that could be followed consistently four nests were produced with a total of 14 eggs in clutches ranging from three to five. Nine eggs hatched–at least one from each nest but all but two ducklings died within a few days of hatching. The two ducklings that have survived were both from the same clutch and ironically were at a site were it was virtually impossible to get any decent observations on their behaviour.

Although the cause of the ducklings deaths can not be known for sure, it is likely to be a combination of a very dry season on the island and poor parenting. Many of the muddy areas in the release area probably dried up reducing access to suitable food. While the least attentive mother, who lost four ducklings, would leave her young unattended for up to 30 minutes at a time. It is also possible that the two ducklings from one clutch were lost due to aggression for a female, which had previously lost her own duckling.

Campbell Island teal are an excellent species for education on island ecology and endangered species and the captive population is now at the point where birds can be released to selected external institutions for display and education. While birds from these institutions will not be used for the initial releases into the wild, to reduce disease risk and maximise control over genetics, they will be permitted to breed the birds to maintain the captive population in case it is required in the future. While there has been some pressure to also maintain a captive population of Auckland Island teal, it is believed that this can not be justified as they are at no immediate risk in the wild and there is the potential of hybridisation with Campbell Island teal.

As discussed above there was little option but to take some teal from Dent if a population was to be established at another site. It was important that as much as possible had been learnt form analogue species prior to attempting the captive breeding program so as to maximise the chances of success. With this species we were lucky in part due to their longevity (the two males caught as adults in 1984 died of old age in 1999).

The future of Campbell Island.

The aim of returning the teal to Campbell Island now appears to be a real possibility in the near future, with the eradication of rats likely to be carried out in the winter of 2001. After several extensive searches it appears that the cats on Campbell Island have died out naturally making the restoration of the island much easier. While eradicating any rodent from an island the size of Campbell is by no means straight forward, the current technology and level of expertise means that, weather permitting, we could be in a position to release teal onto the island as soon as 2003.

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Summary of kaki (black stilt) releases in New Zealand

Richard Maloney & Dave Murray

Introduction

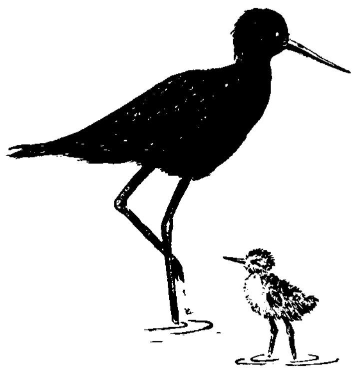
Kaki *Himantopus novaezelandiae* are one of New Zealand's most endangered species, and are one of the rarest wading birds in the world. The total wild adult population is 31, of which only 9-10 are females. Intensive management by the former New Zealand Wildlife Service, and since 1986 the Department of Conservation, has prevent extinction of the species, but the population has remained at less than 50 adults in the wild and less than 15 breeding pairs since 1981. Like many New Zealand bird species, the decline in range and numbers of kaki has been attributed to predation and habitat loss. When Europeans first arrived in New Zealand, kaki were widespread in braided rivers and wetlands throughout the country, but since the 1950s breeding has been restricted to the Upper Waitaki Basin, central South Island (Pierce, 1984 & 1986).

Kaki have been present in New Zealand for about one million years and represent an early invasion of stilts from Australia. Pied stilts *Himantopus himantopus leucocephalus* arrived within the last few hundred years and have quickly adapted to the newly formed agricultural land, and mammalian predator suite that followed European colonization. Kaki differ from pied stilts in morphology, plumage, behavior, mtDNA, voice and in analyses of proteins, and by concordance (Wallis, 1999), and are a separate species. Kaki will hybridize with pied stilts, but most cross-species pairs are male kaki with female pied stilts, and form because there is a severe imbalance of males over females in the kaki population. The mixed pairs that result have fertile offspring, but hybrid fitness is low; survival to adult age is about 50 % of that of kaki pairs.

Intensive management of kaki began in 1981. Since that time the recovery program has focused on protection of breeding pairs and offspring in the wild by (a) reducing predator densities around nesting sites, (b) protection and enhancement of small wetland sites, (c) artificial incubation of eggs before returning them to the nest, (d) cross-fostering young to pied stilt parents, (e) captive breeding, and (f) captive-rearing for release (Pierce, 1996 & Reed, *et al.*, 1993). Usually, one or more of these management actions have been undertaken consistently for up to four years, until outcomes (numbers of kaki reaching breeding age) can be measured. Because kaki breed when 2-3 years old, the recovery program has moved in 3-4 year steps, with new directions or refinements being introduced and unsuccessful techniques (e.g., cross-fostering) stopped at these re-evaluation points (Reed, 1998).

Captive breeding
Successful captive breeding is an insurance against extinction in the wild, and it allows the possibility of releasing surplus stock once a self sustaining captive population has been formed. Captive pairs were first established in 1979 at Mt Bruce Wildlife Center in Wairarapa from eggs taken from the wild. Captive operations were shifted to Twizel in the Upper Waitaki Basin in 1986, and pairs are now held in three localities. Before 1996 breeding was sporadic by all captive pairs, but better pair formation techniques (by flock-mating) and changes in diet at the beginning of each breeding season has resulted in up to 6 pairs producing eggs regularly. Hatching rates of captive eggs have been <30%, compared to 95% for wild eggs, mainly because up to 50% of eggs were infertile, and of the fertile eggs around 50% died while hatching.

In 1999 hatching rates were improved by adding iodine to the captive diet and this season 94% of fertile eggs hatched. Infertility of eggs is probably also related to deficiencies in the captive diet, and this is being researched at the present time. The captive population is now capable of self-maintenance. There are 18 captive adults, and six active breeding pairs, that produced 33 chicks from 61 captive-laid eggs this season. Losses of chicks and juveniles in captivity in the first year average 10 %. Two to four juveniles are held back each year to provide replacement stock for adults, and to form new pairs. Therefore, in the 1999 season there was a surplus of about 25 chicks from captive-laid eggs.



Kaki *Himantopus novaezelandiae*
© E. Murray

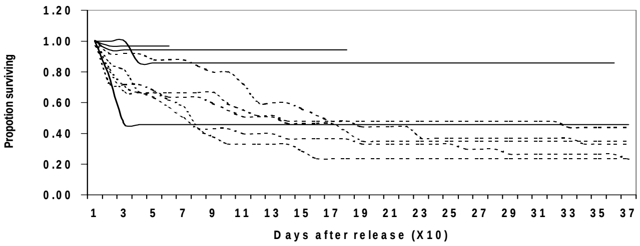
Captive-rearing for release
Releases of low numbers of surplus birds from the captive breeding facility had been undertaken in 1981 (eight juveniles), and from 1987 to 1992 when a total of 22 sub-adults and adults were released. None of the 1981 released juveniles survived the release, but at least four of the birds released up to 1992 reached adult age, paired with wild birds and successfully bred.

With the failure of recovery techniques used up to 1992 to significantly increase the wild population, the emphasis was moved from returning eggs collected from wild pairs back to the wild as hatching eggs, to raising chicks in captivity until they were nine months of age. This avoided a critical period of loss of chicks in the wild (from hatching to fledging). Most of the wild and captive egg production was retained and hatched in captivity in 1992, the chicks were kept and raised by hand, or placed under captive pairs for "parent-raising". The first major release of sub-adults was in September of the following year (1993) when the birds reached nine months of age. This age was chosen because it is the time when wild parents naturally break up from young prior to the next breeding season, and it is after the cold winter period. At first, stilts were released directly from the rearing aviaries into adjoining wetland habitat. All had radio-transmitters attached and each bird was located on each day. Mortality rates were high in the first eight weeks, but after that time few birds died and most birds present at the end of summer survived the winter, and reached breeding age (Fig. 1).

Most deaths were attributed to predation, striking power wires and other trauma-related causes (collisions). Few bodies were collected because predators either killed and ate, or scavenged dead birds before they could be recovered. Partial necropsies of those recovered revealed few health problems, other than the poor body condition of some birds (but some conditions such as goitre were not searched for—see below).

Releases were repeated at this site in 1994 and 1995 (Table 1) despite the high losses of birds hitting nearby wires, principally because survival rates were still 25-45% and this was considered a good result, and because we were unsure of a method of release that would work at remote sites without the need for large holding aviaries. In 1996, a trial release was attempted at a river delta away from pylons. At this site, birds were simply transported to the site in boxes, and released on the same day. Survival rates were similar to previous years with deaths attributed to predators, and some to trauma. Releases in 1997 were also at this site. By now, stilts from earlier releases had formed pairs with each other, or with other wild birds, and had successfully produced young, and appeared to have appropriate behaviors, towards predators, and in choice of nesting and feeding sites. This increased our confidence in the release technique as a method of successfully

Fig. 1. Proportion of stilts surviving up to one year after release. (Dotted lines are stilts released from 1993 to 1997, prior to supplementation with iodine and post-release provision of food. Solid lines are releases from 1998 to 2000, when supplementation occurred).



establishing stilts in the wild.

In 1998 a new release site, 48 km north east of the previous site was chosen. Sub-adults were released at the end of August 1998, and following a snowstorm three days after release 11 of the 15 birds released died within the next seven days. Importantly, most of these bodies were recovered before being taken by predators, and while the snowstorm may have hastened the death of these birds, full necropsies revealed that all 11 birds had thyroid hyperplasia (goitre), which can be caused by a deficiency or excess of iodine in their diet. The presence of goitre was suspected in these birds in 1998 (but not previously) because captive eggs that failed to hatch were found to show all the symptoms of an iodine deficiency that probably related to the absence of iodine in the diet of all captive birds. Goitre most likely affected stilts by lowering metabolic rate, which may have made them more susceptible to death by predation or hypothermia.

The remaining four stilts from this release and a further six birds released that year were all given both iodine and supplementary food after release. Supplementary food was the same food they were given in captivity, placed on plates and left near to feeding areas at the release site. The released birds quickly learned to feed from the plates. Food was provided *ad libitum*, for up to one month after the release, by which time most birds had stopped using the food, and were feeding themselves. After the provision of iodine and supplementary food, no further birds died in 1998, and of the 64 birds that have been released in 1999 and 2000 only two (3%) have been found dead.

The use of iodine and post-release supplementary food is confounded, because both treatments were applied to the released birds at the same time. We do not intend to separate the two factors to determine which may have increased survival rates of released stilts. Both are easily applied, and therefore, we will continue to provide both treatments in all future releases.

Future releases

In total, since 1993, 212 stilts have now been released: four adults, 165 sub-adults and 43 juveniles. Future releases will take place annually, with sites selected on the basis of numbers of adults already in the area, and numbers of released birds surviving in the present location. Up to six release-sites (including the Ohau and Cass sites that have had releases already) will be used. Pulse-releases of large numbers of birds at one site before moving to another is preferred, because it maximizes the number of potential mates each kaki can find.

As February releases of juvenile stilts have been initially successful, releases will continue in both autumn and spring. The potential to release large numbers of birds is restricted by brooder and aviary spaces, by staff time, and by the number of eggs laid by captive and wild pairs. About 70 birds per year can be produced at a maximum. With the present high survival rates following addition of iodine to the stilts captive diet, and post-release supplementary feeding, the opportunity to increase the wild population to more than 150 birds within the next few years is very real.

However, releases will only serve to increase present population numbers, and do not address causal factors that limit the ability of wild stilts to breed successfully in the wild. Research into methods of mitigating factors influencing breeding and recruitment in the wild continues. These are mainly focused on identifying causes of chick and adult female mortality, and on improving methods of

Table 1. Site, number, timing, method and success of releases of juvenile and sub-adult stilts reared in the Twizel Aviary. Four adults released in 1999 or 2000 have been excluded from these totals. Methods of release are: A = released directly from rearing aviaries, not given supplements; B = caught, transported and hard released on the same day away from the rearing aviaries, not given supplements; C = as for B, except given iodine in pre-release diet and given supplementary food for up to one month after release. In addition all birds in all years were screened for presence of parasites and dosed as required, prior to release.

SITE	YEARS	TIMING	AGE OF RELEASED BIRDS	NUMBER OF RELEASES	NUMBER OF BIRDS RELEASED	% SURVIVAL TO BREEDING AGE	METHOD OF RELEASE
Twizel aviary	1993,1994 & 1995	September (spring)	Sub-adult (9 months)	3	79	28	A
Ruataniwha wetland	1996	September (spring)	Sub-adult (9 months)	2	7	29	A
Ohau River Delta	1996 & 1997	September (spring)	Sub-adult (9 months)	4	41	39	B
Cass River	1998	September (spring)	Sub-adult (9 months)	2	21	48	B, C
Cass River	1999	February (Autumn)	Juvenile (3 months)	1	10	100 *	C
Cass River	1999	September (spring)	Sub-adult (9 months)	1	17	N/A	C
Cass River	2000	January & February (autumn)	Juvenile (3 months)	2	27	N/A	C

*** None of these 10 juveniles have been found dead within the first one year after release ~ they will not be 2 years old until after September 2000**

controlling predators.

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Translocation history of hihi (stitchbird), an endemic New Zealand honeyeater

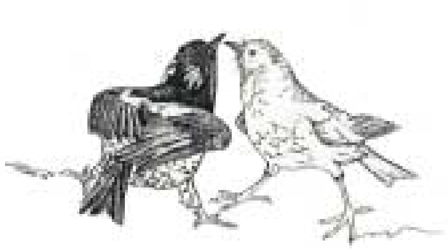
Shaarina Boyd & Isabel Castro

Introduction
The hihi *Notiomystis cincta*, or stitchbird is the rarest of New Zealand's nectar feeding 'honeyeaters'. Hihi are a member of the Meliphagidae family that is wide spread in Australasia but only represented by three species in New Zealand. A medium-sized (30-40g) forest bird, hihi are also one of the rarest species of this family in the world.

The male sports a flashy plumage of black head with white 'ear' tufts, bright yellow shoulder and breast bands, a white wing bar and a mottled tan/grey brown body cover. The female is more subdued with olive-grey brown body cover, white wing bars and small white 'ear' tufts. Hihi are at the bottom of the pecking order when competing with their close relatives the tui *Prosthemadera novaeseelandiae*, the largest and most dominant of New Zealand honeyeaters) and bellbirds *Anthornis melanura* for food resources such as nectar, and fruits (Craig *et.al.*, 1981 & Rasch, 1989). Hihi are the only honeyeater in the world that is known to have a variable mating system and one of two species that nest in cavities. They are also the only birds known to sometimes mate face to face.

In pre-European times, hihi, were found throughout the North Island mainland and on Great Barrier, Little Barrier and Kapiti Islands. The species became extinct on the mainland with the last recorded sighting in the Tararua Ranges in 1883. The only surviving population was found on Little Barrier Island (3,076 ha) in the Hauraki Gulf which was declared as a Bird Sanctuary in 1894 and subsequently a Nature Reserve. The formation of this sanctuary undoubtedly saved the hihi from extinction. The disappearance of hihi from the mainland coincided with the introduction of cats and ship rats to New Zealand, extensive forest clearance and probable introduction of avian diseases

Early conservation efforts
The presence of only a single population on Little Barrier Island was deemed to be vulnerable and hence the New Zealand Wildlife Service initiated and established a program of translocations to cat and ship rat free islands between 1980–1987. The translocation program has been continued by the New Zealand Department of Conservation and now forms part of a Hihi Recovery Program (Rasch *et al.*, 1996). Offshore island sanctuaries are critical to the survival of hihi. They have been the only ship rat-free environments left in New Zealand until recent advances in pest control and the establishment of New Zealand's "Mainland Islands" program.


Stitchbird (Hihi) ***Notiomystis cincta***
© Liz Grant

Since 1980 there have been a total of 12 translocations of hihi to five different islands. The Wildlife Service focused their efforts on Hen (718 ha; two transfers, 46 birds total), Cuvier (181 ha; two transfers, 66 birds total) and Kapiti Island (1963 ha; three transfers, 60 birds total). Due to the remoteness of Hen and Cuvier Islands, minimal monitoring was undertaken. However this monitoring showed that after translocation the population declined to extinction on Cuvier and only 1—2 birds on Hen.

Translocation development
The first draft of the Hihi Recovery Plan focused the translocation efforts toward a more rigorous monitoring and experimental phase. Between 1990–1992, three translocations to Kapiti Island totaling 107 birds were undertaken. Trials of hard (release directly from transfer boxes to the wild) versus soft release (on-site aviaries with birds released two weeks after initial hard release groups), and pair versus group releases were undertaken in 1991. This revealed better survival when a hard, group release technique was used. A high level of aggressive social interactions was observed between the more established hard release birds and the aviary held birds (Castro *et al.*, 1994). This led to tests in 1992 of the survivorship of birds released in the presence and absence of conspecifics. This showed that hihi survived better when released in areas free of conspecifics. Concurrently supplementary nectar feeders and artificial nest boxes were made available to support monitoring and assist establishment. Hihi used feeders only during the breeding season. Nest boxes were used in addition to natural cavities. By 1994 a small population of approximately 40 birds was established on Kapiti and remains stable through to 1999.

The knowledge gained on translocation and release techniques, nest box designs and supplementary feeder regimes as tools to enhance establishment, allowed for the consideration of smaller islands with regenerating forest as potential translocation sites. In 1994 a translocation of 40 birds to Mokoia Island (135 ha) provided opportunities to investigate the effect of food supply and supplementation on the survival (Armstrong *et al.*, 1999), breeding system and establishment of hihi. The research obtained clear evidence that high mortality rates detected in the non-breeding season were not due to food limitation. However results on the influence of food supply on mortality in the breeding season were variable and unclear. The absence of bellbirds on Mokoia Island offered another opportunity to compare the survival of hihi released in the absence of this competitor with that of hihi released into a complete honeyeater guild on Tiritiri Matangi Island.

In 1995–96, two translocations of a total of 51 hihi were made to Tiritiri Matangi Island (222 ha). Investigations of honeyeater interactions at both sites reinforced the knowledge of interspecific competition for food resources between the three honeyeater species. However these interactions did not appear to adversely affect hihi survival and establishment. Additionally, the same supplementary feeding experiments were repeated on Tiritiri Matangi Island as on Mokoia Island. On Kapiti and Mokoia Islands feeder use was heaviest during early breeding however on Tiritiri Matangi Island heaviest use occurred during final chick rearing phases. At other times feeder use was minimal. However local variation in flowering and fruiting abundance between years does appear to influence productivity of hihi at all sites.

Active management

Management techniques to assist hihi productivity and survival have been developed. These include:

- the design and provision of nest boxes allowing access for monitoring and nest manipulations.
- provision of feeders to enhance food supply and monitoring success.
- banding of all fledglings for individual identification and measurement of demographic parameters.
- intensive management of mite infestations at nests.
- candling of eggs to determine development
- cross-fostering of eggs and chicks where necessary.

Management on Kapiti Island has been limited to provision of feeders during the breeding season, baseline monitoring, and the eradication of "kiore" *Rattus exulans* and Norway rats *Rattus norvegicus* rats in 1995. Intensive management has occurred on both Mokoia (1994–1998, 2000) and Tiritiri Matangi Islands (1997–2000).

A small captive population is held at the Mt Bruce Wildlife Centre. They have contributed to species advocacy, understanding of disease issues, developing techniques on captive husbandry and also provided birds for experimental release to the wild. Four birds have been released (on Tiritiri Matangi Island). Two survived, one of which entered the breeding population in 1999/2000.

Research

Research programs by Massey and Auckland Universities have greatly increased our knowledge of hihi. Researchers and wildlife managers have contributed to developing suitable management programs.

Research focus areas

Habitat use: In assessing new sites current information indicates that year round nectar and fruit sources are important to sustain hihi. 'Gaps' in nectar or fruit availability at critical times have been identified by increased use of supplementary food. Invertebrate foraging occurs year round but is most prominent during chick rearing. Overlap in food requirements is greatest with the similar sized bellbird. Male bellbirds in particular are the biggest competitor at concentrated food sources. Hihi are mobile, living in overlapping home ranges, only constrained to territory areas during breeding, which improves the species chances to access food.

Social interactions: A linear hierarchy exists between the three honeyeaters with tui dominating bellbirds who dominate hihi. Resource partitioning occurs at dispersed food resources while strong territorial defense and high levels of interaction occur at concentrated food resources. Hihi display territorial behavior at nest sites and concentrated food resources.

Mating systems and parentage: Hihi have a variable mating system with males competing for copulations. The birds breed monogamously or in groups (polygyny, polyandry, or polygynandry) depending on the distribution of available nesting cavities. If nest sites are far from each other, one male will mate with a single female. However if nest sites are close together, a single male or two males may pair with several females in the same area. Two males and a female have been observed nesting together when the number of males exceeds the number of fertile females available. Males also try to achieve paternity by means of forced copulation, adopting a face to face posture during mating (Castro *et al.*, 1996). DNA finger printing has revealed chicks in a nest may be fathered by more than one male. Fathers may include the resident male, and 'paired' or unpaired males (Ewen *et al.*, 1999). Supplementary feeding at nest sites appears to influence male contribution to chick rearing at the nest site.

Disease identification and management: Aspergillosis, a fungal infection of the respiratory system, has been implicated as a significant cause of mortality on Mokoia and Kapiti Islands. This infection is not usually considered a primary disease, but in other species has been associated with immuno-suppression due to stress e.g. translocation, crowding, other diseases. Aspergillosis is not thought to be contagious. However, its continued occurrence in the population has raised the possibility that the hihi's immune system may be challenged by social stress, hormones or an interaction between these factors, or that Aspergillosis is indeed contagious. Identification of environmental factors and transmission vectors associated with this disease are currently being investigated.

Population viability analysis: Using the Vortex software and parameters from the Mokoia and Tiritiri Matangi Island populations have been used. This preliminary work indicates that the viability of the Tiritiri Matangi Island population appears good, but that the Mokoia Island situation is unstable and would depend upon the continued management of nest mites.

Current status of hihi

Little Barrier Island is still the only self-sustaining unmanaged population. Currently all the remaining translocated populations consist of approximately 40 birds, giving a total of 120 hihi outside of Little Barrier Island. Although all transfer populations continue to breed and recruit, only one population, Tiritiri Matangi Island appears to be expanding with management assistance. However none of the translocated sites are self sustaining.

A desperate lack of information on the status of the Little Barrier Island population exists and efforts to acquire suitable funding to undertake this work continue. Intensive research is required here as baseline information on demographic parameters, and disease occurrence is needed.

Conclusions

- Hihi survive translocation better if hard released into areas where there are no conspecifics present.
- When assessing new sites current information indicates the continuous presence of flowering and fruiting species is a

prerequisite for hihi establishment.

- The use of supplementary nectar feeders can successfully bridge resource gaps.
- The provision of specially designed artificial nest boxes in the wild can be successful in supporting hihi breeding. This allows the eligibility of regenerating habitats as translocation sites.
- Management of nest mite infestations in artificial nest boxes is advisable.
- Both feeders and nest boxes greatly improve options for assisted management, research and monitoring.
- Hihi are subordinate to other New Zealand honeyeaters and compete most closely with bellbirds for resources.
- Competition for food between male and female hihi is as intense as interspecific competition with bellbirds. This places female hihi at the very bottom of the hierarchy and subject to high levels of social stress.
- Susceptibility of hihi to aspergillosis and coccidiosis has potential to be intensified by social stress and/or hormonal responses.
- Resource distribution can be used to manipulate hihi breeding strategy and could be developed to optimize productivity.
- Captive reared and released progeny can be successfully recruited into the wild breeding population.

Despite all this knowledge and a range of suitable management techniques, hihi still only have a single wild, unmanaged, self-sustaining population on Little Barrier Island. Translocations have assisted in developing our knowledge of the species biology and management, but translocated populations remain shaky. The next phase of the Recovery Program will be to continue to support the transfer populations through co-coordinated management, and to return to Little Barrier Island to obtain critical information on demography and disease. This information will allow us to assess the relevance of the patterns found in the translocated populations.

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Fifty years of conservation management and re-introductions of the takahe in New Zealand

Ian Jamieson, William Lee & James Maxwell

Background

The unexpected discovery in 1948 of a population of takahe *Porphyrio hochstetteri* (formerly *mantelli*), a large, flightless, endemic rail, in Fiordland on the South Island of New Zealand, could be considered one of the great ornithological discoveries of the past century. Known only from a few early European sightings, the takahe had not been seen since 1898 and was generally considered to be extinct. The finding of a population of takahe initiated a major conservation effort to protect the species, which has continued uninterrupted for 50 years. Research and management efforts to save the takahe have been unprecedented in the history of threatened species conservation in New Zealand. Initially these involved establishing a special 503 km^2 area for the protection of takahe centered on the Murchison Mountains within Fiordland National Park. Efforts have been greatly extended subsequently to include a range of techniques and approaches including captive-rearing for release into former habitat, augmentation of the remnant population, and translocations to predator-free islands.

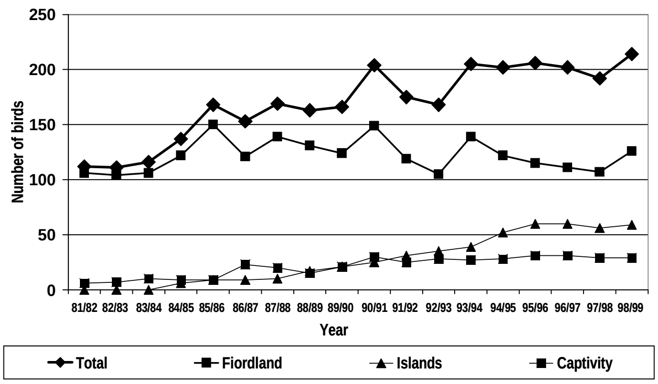
The first censuses of the Fiordland population in the 1960s put the number of adult birds at approximately 250. By 1981, there were about 106 adult birds. Currently (1999) there are at least 210 adult takahe, but only about 120 remain in Fiordland, the rest being primarily on 4 offshore islands or in the Burwood Captive Rearing Unit near Te Anau (Fig. 1). The growth in takahe numbers over the last 20 years has been largely due to the increase in birds on islands, 75% of which are now island-bred birds. The original Fiordland population is largely centered in the Murchison Mountains, and now includes at least 26 captive-reared birds. Intensive management efforts with the Fiordland population have halted the decline in number of birds, which was evident by the mid-1970s, but the numbers have not increased. Part of the reasons for this lie with the re-introduction program.

Early research before onset of re-introduction program

One of the first priorities of the takahe recovery program was to identify the cause of the decline and the life history stage when birds were most at risk (Bunin & Jamieson, 1995). One of the major findings of the earlier research was that introduced red deer *Cervus elaphus scoticus* and takahe preferred the same plant-food (alpine tussocks), but that deer grazing retarded the growth and vigour of the tussocks for several decades (Lee *et al.*, 1988). Improving habitat quality in Fiordland became an important focus for the conservation of the takahe during this period, resulting in the introduction of intensive deer control.

The second major research finding was that takahe chicks had a less than a 30% chance of survival in their first year. This was thought to result from competition for food with deer, predation by introduced mammalian predators, and the harsh alpine weather.

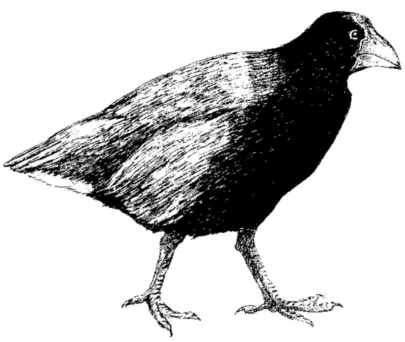
Fig. 1. Total number of takahe and numbers in Fiordland, islands and captivity



In addition, takahe rarely raised more than one chick, yet clutch sizes of two or three eggs are common. Artificial incubation of the 'surplus' eggs was undertaken, eventually leading to the development of a specialist captive-rearing center near Te Anau. For over 10 years now, between 10–16 fertile eggs have been taken from the wild birds in Fiordland each spring, artificially incubated, and the chicks raised using a combination of techniques including hand puppets and natural grassland enclosures, with a high success rate (90% hatching and 75% fledging). The regular availability of captive–reared birds transformed the conservation management of takahe, enabling the establishment of new populations on offshore islands (see below), but most were returned to Fiordland as yearlings (Bunin & Jamieson, 1995).

Re-introduction to Stuart Mountains

From 1987 to 1992, 58 captive-reared yearling takahe were released in the Stuart Mountains, a former part of their range to the north of the Murchison Mountains, in an attempt to initiate a new population that would eventually merge with the wild birds. By 1993, 22% of the released birds were known to have survived their first year in the wild, with most birds (72%) unaccounted for. One pair was known to have bred successfully and by 1994, only eight takahe (14%) were sighted in the area. The release program for the Stuart Mountains was stopped in 1993 due to concern over the difficulties with monitoring the population and the apparent low number of birds remaining in the release region. At the same time the takahe population in the Murchison Mountains was declining, and priority was shifted to using captive–reared birds to boost numbers there. Reasons why the release program apparently failed are unclear, although several possible factors may have contributed. Habitat quality of the Stuart Mts. may have been lower than previously considered as deer densities remained high there compared to the Murchison Mts. Establishment may have been assisted if a larger number of birds had been released at a time, or if birds had been released when they were slightly older (two years) and had already formed pairs. The Stuart Mts. also lacked a resident takahe population, which if present would probably have encouraged the formation of settled, breeding pairs, and possibly would have assisted released birds to learn survival skills. Finally, pre-release training of some birds may have been inadequate-half of those released in the Stuart Mts. were reared in the earlier stage of the captive rearing program, before the practice of teaching birds to forage for *Hypolepis* rhizome, important winter food for takahe, was developed.



Takahe *Porphyrio hochstetteri*

Augmentation of the Murchison Mountain population

Since 1991, captive-reared takahe have been returned to the Murchison Mountains to assist the rebuilding of the original population. Here the release program has been much more successful. At least 60% of the released birds have survived their first year in the wild and they now make up 26% of the total population with over 30% of all breeding pairs involving at least one captive-reared bird (Maxwell & Jamieson, 1997). Yet, despite the high recruitment of captive-reared birds, the overall population in the Murchison Mts. has not increased (Fig. 1). One of the main reasons for this is that 5 of the 7 release years have coincided with unusually cold winters, which are known to negatively affect recruitment (Maxwell & Jamieson, 1997).

Although the number of captive-reared birds released at age one year (N = 86) is almost twice that of wild-reared birds of the same age (N = 45), the two groups have produced the same number of yearling offspring. It is possible that captive-reared birds are significantly less productive than their wild-reared counterparts, to the extent that any increase in recruitment at the juvenile stage through captive-rearing is nullified by lower fecundity. These concerns highlight the importance of continued monitoring of pairing and breeding success to ensure that the management is assisting recruitment of quality breeders.

The impact of continuous egg removal on lifetime reproductive success of individual breeders and on the long–term population dynamics of takahe in Fiordland has yet to be evaluated. The main reasons for this are that (1) a large proportion of the original population in the Murchison Mts. was unmarked, (2) transferring breeding records to a computer database has been slow (because of other priorities) and (3) a lack of expertise in population modeling. These outstanding issues will need to be addressed before the success of the captive-rearing and release program can be fully assessed. Evaluating the impact of specific management actions will still be difficult because cause and effect are often difficult to determine when multiple actions (e.g. deer control, transferring viable eggs between breeding pairs, collection of eggs for captive–rearing) are being undertaken. Adaptive management, involving the testing of hypotheses using experimentally designed management programs, should be the way we proceed in the future.

Introductions to predator–free islands

Takahe have also been successfully introduced on offshore islands which have non-native pasture plant species but are free of mammalian predators. Between 1984 and 1993, a total of 24 takahe (mostly captive–reared yearlings) were released on four islands which were managed as a single population. As of 1999, there are 59 yearlings and adult birds and the island population as a whole now comprises over 25% of the total takahe population. Recruitment of juveniles is high on islands primarily because of the lack of mammalian predators and the benign weather conditions relative to Fiordland. However, island breeders produced twice as many infertile eggs as Fiordland birds (Jamieson & Ryan, 2000). The high rate of egg infertility is thought to be due to inbreeding depression associated with translocating inbred individuals (from the relict Fiordland population) to a habitat (i.e. pasture grassland) in which takahe have had no evolutionary history. The poor hatching success may not have long term implications for the survival of these birds on islands as long as recruitment remains high (Jamieson & Ryan, 2000).

Conclusions

Have management efforts including captive rearing for release back into the wild contributed to the conservation of takahe? In our view, the takahe would be extinct in the wild without the employment of these management tools because of the small population size, predation by introduced mammals, and extremes of climate. Currently, the threat of extinction of the species has been reduced. However, the challenge for the next 50 years is to develop viable populations in several localities on the mainland and on more offshore islands. This will require an ongoing commitment to research and management, and the development of new technologies, especially if the birds are to remain in Fiordland where they were re-discovered just over 50 years ago.

Most of the information presented in this report has been extracted from a forthcoming book on the proceedings of a symposium entitled, "The takahe: 50 years of conservation management and research", held at the University of Otago in November 1998 (Lee & Jamieson, in press). The book not only discusses in detail the successes of the Takahe Recovery Program, but also dissects some of the failures as well as questioning the overall approach to conservation management of endangered birds in New Zealand. If it is possible to learn from the success and failures of others, then this publication should be of interest to other avian re-introduction programs.

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Re-introduction of orange-bellied parrots, Australia

Ian Smales et al.

The orange-bellied parrot *Neophema chrysogaster* is one of a number of bird species that annually migrate between wintering grounds on the Australian mainland and breeding areas in the southern, island State of Tasmania. The parrot is critically endangered, with a wild population believed to number less than 200 individuals, which breed in a very restricted range.

The bird is one of six small (40-50g) parrots forming the genus *Neophema*, which spend much time feeding on, or close to the ground. Orange-bellied parrots breed in the hollows of eucalyptus trees fringing moorland plains in the remote, cold and relatively unspoilt Wilderness World Heritage Area of Tasmania's south–west. Here they feed upon the seeds of sedges, grasses and other plants of the plains. By contrast, they over-winter in salt-marshes and coastal dune vegetation on the coast of Victoria and South Australia, which have been subject to massive loss and degradation caused by human activities during the past 200 years. Their migrations involve two annual ocean crossings, each totaling more than 200 km, and with a longest single passage of about 90 km of open water.

In recent decades the known over-wintering groups have mostly concentrated in small areas of suitable habitat on the western coast of Port Phillip Bay near Melbourne. These sites have been threatened by various industrial development proposals. The resultant campaigns and recovery efforts to conserve the parrot have drawn public attention to its plight and, for the present, prevented several potentially adverse developments from proceeding.

Loss of critical winter habitat is considered to be the primary cause of both the decline of the orange-bellied parrot and of present limits on the population (Menkhorst *et al.*, 1990). The breeding range in Tasmania has also contracted considerably, but reasons for this are not clear.

A wide range of activities aimed at recovering the population and conserving its habitats are outlined in a recovery plan (Orange-bellied Parrot Recovery Team, 1999). A recovery team representing a number of organizations, including Birds Australia and agencies of the State Governments of Tasmania, Victoria, and South Australia facilitates implementation of the plan. During the past 15 years these measures have prevented further decline in the number of breeding birds in the wild, and permitted a small population growth in recent years (Orange-bellied Parrot Recovery Team, 1999).

Captive management for conservation of orange-bellied parrots commenced in 1986 following successful trials of captive management and release of closely related rock parrots *Neophema petrophila* and blue-winged parrots *Neophema chrysostoma*. From then until early 2000 more than 400 orange-bellied parrots have been bred and raised in facilities operated by the Tasmanian National Parks and Wildlife Service and by Healesville Sanctuary, in Victoria. Since 1991, 105 captive-bred birds have been released into the wild.

Re-introduction

Between 1991 and 1999, seven releases of captive bred birds have been conducted at two wild breeding sites in coastal southwest Tasmania (Brown *et al.*, 1994). A total of 99 birds has been released there to date. Numbers of birds released, their year and location of release, and subsequent sightings are shown in Table 1.

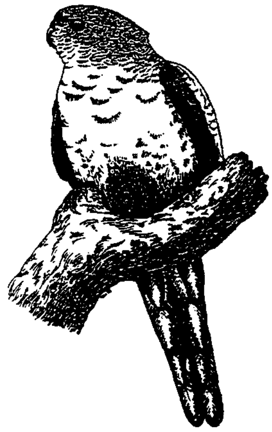
Releases have been successfully carried out at Melaleuca, the main natural breeding site, in Tasmania's south west, and at Birch's Inlet, a former breeding location, further to the north. Cabin accommodation at both locations allows a roster of volunteer observers to work throughout each breeding season. An observatory has been built at Melaleuca for the purpose of monitoring the wild population and released birds. At each of these locations, parrots have been released from an aviary built in the natural sedgeland feeding habitat of orange–bellied parrots.

In 1991 a mixture of adults and young birds hatched during the previous breeding season, were released. In subsequent years, mostly first-year birds have been liberated, due to their potential reproductive life. Parrots for release have been acclimatized to the location by maintaining them in the aviary for up to a month prior to release. Early in the breeding season, after the wild birds have arrived from the mainland, a panel of small mesh in the upper section of the aviary is replaced with larger mesh. This permits the parrots passage in and out of the aviary, but prevents larger predators from entering the cage. For the first few days birds have flown to and from the aviary and mingled with wild birds. Some individuals have been observed to return to the aviary to roost for a few nights.

Colored leg bands are fitted to released birds for identification and to denote their year of its release. A feeder table at each of the two sites is supplied with seed. This provides a focal point at which orange–bellied parrots can be observed and identified. Because the birds have short legs and spend much time in dense vegetation on or close to the ground, accurate observation of leg bands other than on the feed tables is almost impossible. Observations of birds attending the tables indicate that the great majority of released birds survive well, at least during the breeding season of their release. At both locations nesting boxes have been erected in trees in the release area. Most released birds quickly form partnerships amongst themselves or with wild birds and some are known to have bred immediately after release. Many of them have thus contributed genetically to the wild population.

Table 1. Annual numbers of orange-bellied parrots released to wild locations and subsequent re-sightings

	RELEASE SITES			RE-SIGHTINGS		
	Melaleuca	Birch's Inlet	Pt. Wilson (mainland)	Melaleuca	Birch's Inlet	Mainland (various locations)
1991	10	-	-	-	-	-
1992	15	-	-	-	-	3
1993	14	-	-	2	-	1
1994	-	14	-	1	-	-
1995	-	-	-	-	-	1
1996	-	13	6	-	-	-
1997	-	-	-	-	1	-
1998	-	-	-	-	1	-
1999	-	33	-	-	-	-
	39	60	6	3	2	5



Orange-bellied parrot
Neophema chrysogaster

Five individuals released in the course of the program have been sighted during subsequent winters on the Victorian coast, one in two consecutive years. Four have been sighted at the breeding areas in subsequent years. One such was a bird released at Birch's Inlet, which reappeared there in February of 1997, after an absence of more than two years. The number of re-sightings of released birds is small, but is consistent with a low rate of identifications of all wild orange–bellied parrots. This occurs because the birds feed amongst vegetation which obscures their leg bands.

Despite releases, and breeding in the wild by released birds, the total population grew little between 1991 and 1996. The numbers of birds released may have been insufficient to make a difference to overall population size. In order to permit growth in the captive population, no birds were re-introduced in 1997 and 1998. In September 1999, following a very productive season in captivity, 31 first-year birds were liberated at Birch's Inlet. In the absence of a wild population at that locality, a breeding pair of second–year birds were kept in the aviary as 'call birds' to encourage the released birds to remain in the vicinity. Following the breeding season the pair were also released. At least seven juveniles are known to have been produced by three pairs that formed following the release in September. The Birch's Inlet area will be monitored to determine whether released birds return in future years to breed.

The 1999 re-introduction constituted more than twice the number of parrots released in any previous year. It is hoped that releases of this magnitude will be able to be repeated in future years and that they will begin to significantly boost the size of the wild population.

The recovery team has also undertaken one trial of the release of captive–bred birds into mainland winter habitat. Objectives of the trial were to determine if birds could survive such a release and also to see whether birds were capable of undertaking the southward migration to Tasmania in spring.

An increased availability of captive–bred birds, following establishment of the Healesville Sanctuary colony, allowed this trial to take place during the winter of 1996 (Menkhorst, 1997). Three juvenile males and four juvenile females were selected for release. An aviary was erected on the beach edge in saltmarsh habitat close to a traditional wintering site for wild orange–bellied parrots at Point Wilson, on the western side of Port Phillip Bay. Since most of the wild flock return annually to Tasmania in late September, the release was timed for August to allow them a number of weeks to acclimatize to the wild environment and hopefully to integrate into the wild flock and migrate with it. The birds were maintained in the aviary for three weeks prior to release, by staff from Victoria's Open Range Zoo at nearby Werribee.

Cut saltmarsh plants, of varieties that are natural foods of orange–bellied parrots, were put into the aviary regularly and were utilized by the birds. Each bird was individually color–banded. Four individuals were fitted with small radio transmitters, glued to the upper shafts of their two central tail feathers, three days before they were released. This allowed the movements of the group to be monitored, for at least the six weeks expected life of the transmitters. Techniques for attachment had first been trialed on closely related elegant parrots *Neophema elegans* at Healesville Sanctuary. In most other respects the re-introduction process was similar to that employed to release parrots in Tasmania.

One individual died of unknown causes whilst in the aviary, but the other six were freed together in late August. One bird could not be found after six days, but the remaining five were located in various combinations of individuals until 26 days after their release. The last two longest remaining were sighted 39 days after release. Radio telemetry and sightings showed that they adapted readily to wild foods and that they moved up to 3.5 km from the aviary. Some traveled at least to where the wild group was living but it could not be confirmed that interactions took place. The time of their disappearance coincided, within days, with the return of the wild birds to Tasmania, however none of them have been seen since their disappearance from Point Wilson. The trial is considered to have been a qualified success in that the birds readily adapted to the wild environment and survived for a number of weeks (Menkhorst, 1997).

Techniques have been developed and thoroughly proven for successful reintroduction of orange–bellied parrots to the wild. Building on these techniques, the recovery team plans to continue re-introductions as part of efforts to conserve the parrot.

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Re-introduction of helmeted honeyeaters, Australia

Ian Smales et al.

The helmeted honeyeater *Lichenostomus melanops cassidix* has declined to one population now numbering about 100 birds along 5 km of remnant streamside habitat in Yellingbo Nature Conservation Reserve, in southern Victoria. The wild population reached an all–time low of about 60 birds, including just 15 breeding pairs, in 1990.

Since 1989 the helmeted honeyeater has been the subject of an intensive collaborative recovery effort led by the Victorian Department of Natural Resources and Environment, Healesville Sanctuary and a community group, Friends of the Helmeted Honeyeater (Menkhorst & Middleton, 1991).

Study of the natural population since 1984 has shown that breeding birds live in social communities comprised of a number of territorial pairs. These communities inhabit riparian forests of very specific species and structural composition. Pairs of helmeted honeyeaters exhibit long-term fidelity to partners and territories. Non–breeding birds are known to move between communities and to make some explorations beyond the riparian zone. Natural establishment of a helmeted honeyeater community in unoccupied habitat has been recorded only once and that was into an area which birds had vacated just four years previously.

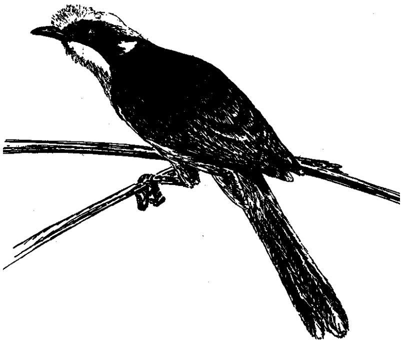
Captive management of helmeted honeyeaters began at Healesville Sanctuary in late 1989 (Smales *et al.*, 1992). The captive population was established by foster-raising helmeted honeyeaters from wild–laid eggs placed under Gippsland yellow-tufted honeyeaters *Lichenostomus melanops gippslandicus*. The resultant captive adult helmeted honeyeaters have been reproducing consistently well since 1995.

Re-introduction experiments

Whilst much of the former range of the helmeted honeyeater has been cleared of natural vegetation, the northern portion is now a mosaic of forest patches of various sizes amongst cleared land. During the latter half of the twentieth century, the decline of the helmeted honeyeater has included the sudden loss of some populations when forest patches they inhabited have been burnt by wildfire. Where this has occurred, regeneration of natural forest communities has often followed so that some substantial areas of habitat now appear quite suitable for helmeted honeyeaters. However, no such areas have been re-colonization by the bird. This may be due to limited dispersal capacity on the part of the dwindling population within a fragmented landscape.

Between 1993 and early 2000, 34 birds have been released from aviaries in the Yellingbo Nature Conservation Reserve in re-introduction trials. The location was inhabited by a natural group of helmeted honeyeaters until 1978. Currently, the closest group of wild birds occurs along a different creek, about 1 km distant and separated from the release site by a low ridge of cleared farmland.

The aim of re-introduction trials is to develop techniques that encourage birds to form fidelity to a site where they will reside and breed. The intention is that successful methods can later be applied to re-establish groups of helmeted honeyeaters more widely within their former range. The concept behind the trials has been to begin with simple techniques and progress towards more complex methods only as necessary. Wherever possible, advantage has been taken of knowledge of the biology and ecology of the bird in the wild and of captive husbandry techniques, in planning and design of trials. Development of re-introduction techniques has been a progressive, experimental process, incorporating numerous variables, amongst which have been:



Helmeted honeyeater
Lichenostomus melanops cassidix

- Origin of birds (translocations of wild birds and release of captive–bred birds)
- Release of birds with or without their having bred in on–site aviaries
- Release of birds of various ages and breeding status
- Variations in group size and social composition of released birds
- Use of decoy birds in attempts to attract wild birds
- Variation in the period birds are acclimatized in on-site aviaries
- Period of supplementary feeding post-release

The following briefly outlines the types of release trials that have been undertaken.

All birds released and offspring they have produced, have been marked with individually unique combinations of colored leg bands. This has permitted accurate monitoring of the birds after their release.

Translocation of wild birds
Initial trials were with translocation of wild birds, involving capture of adults from the wild, holding them for various periods in an aviary at the release site, followed by release. Four birds were translocated and released. A breeding pair were moved, housed in an aviary for six weeks and then released in 1993. Later in 1993 a male and female, both of whom were of breeding age but not paired, were captured and held in the aviary for 10 months prior to release.

Three of these birds left the release location on the day that they were released. All three of them rejoined the wild population and subsequently bred there. The fourth, a male, remained at the release site for almost a month before disappearing.

Release of captive birds
The entire captive group, at Healesville and at the release site, began breeding consistently well during the 1995/96 breeding season and this allowed for the first trials involving captive–reared birds.

Release of captive–bred birds after breeding in on–site aviaries
In this method a pair is maintained until they have bred inside an aviary at the release site and are attending nestlings. The aviary is then opened and supplementary food is provided outside it. The timing of the release is planned to capitalize on the adult birds' strong instinct to keep returning to feed chicks at a time when their familial bond and territoriality are at a peak.

Between late 1995 and early 1999, four pairs has been released by this method. They have been accompanied by five juvenile offspring hatched and reared in nests within aviaries at the release site. Three of the adult pairs have remained in the release area for the rest of the breeding season in which they were released, and have bred and successfully reared nine additional offspring after their releases. However, only one such pair has remained at the site for more than 12 months. The other adult birds have remained up to a few months after the initial breeding season, but have then disappeared and their fate is unknown. Four offspring of released birds, hatched either within release aviaries or in the wild after their parents had been liberated, are known to have survived into adulthood.

Release of potential partners for resident males
In the breeding season of 1997/98 two captive–reared adult females were released after having been maintained for a period of weeks in aviaries at the release site. Both were released as potential partners for single, resident males. The males were the progeny of previous released birds. In the first case, much social interaction was observed between the male and female, through the aviary mesh, prior to the release. This female formed a pair bond with the male and after her release they built nests, but no evidence was found of egg laying. She was last recorded at the site 4.5 months after her release. In the second instance, no pair bond was evident, no subsequent reproductive activity was recorded and the female disappeared from the area three months after being released.

Rapid release of family groups
In the breeding seasons of 1997/98 and 1998/99, three family groups were released following very brief acclimatization at the site. In each case an adult pair, accompanied by advanced, though still dependant fledglings, was transferred from Healesville Sanctuary. Each family was held for one night in a release aviary prior to being liberated. Each of these releases occurred late in the breeding season and none of the pairs bred again in the season after they were released.

The first family was released in this manner early in 1998. The adult male had been captured in the wild during its second year of life, in 1994. His partner had been reared in captivity. This adult pair and the single juvenile released with them remained at the release site for three months before moving to join the wild population on Cockatoo Ck. The two pairs of captive–raised adults released in early 1999 remained for a few weeks before disappearing. However, three of the juveniles released with them have been sighted frequently at the release location and two of them have survived for more than 14 months until the time of writing.

Release of non–breeding adults
A pair of sexually mature, captive–raised birds were maintained in an aviary at the release location throughout the 1998/99 breeding season. This pair did not breed in the aviary in that time and were released at the end of the breeding season. They disappeared four months later.

Summary of results to date
Between 1993 and early 2000, 34 birds (11:13) and 10 juveniles have been released on site.

- Four have been adults (2:2) translocated directly from wild locations to aviaries at the release site.

- Two have been adults (1:1) collected as such from the wild and held in captivity for 42 and 47 months, respectively, prior to release.
- 28 (8:10) and 10 juveniles have been captive–bred or reared.

Survivorship of birds after release

Of 31 Helmeted Honeyeaters released between 1993 and early 1999:

- 27 are known to have survived in the wild for longer than one month
- 23 are known to have survived in the wild for longer than three months
- 16 are known to have survived in the wild for longer than six months
- 11 are known to have survived in the wild for longer than 12 months
- Four are known to have survived in the wild for between 24 and 53 months.

(Individuals released under all of the different experimental scenarios, described above, have survived for periods of six months or longer).

Post-release breeding

- Three pairs have successfully bred in the re-introduction area after their release. All three have been pairs which had successfully bred in aviaries at the site prior to their release. The first pair re-introduced by this technique bred in the season of their release and in the subsequent season.
- These three pairs laid 32 eggs in the wild after they were liberated.
- Nine chicks have been reared to the point of successful fledging by these three pairs in the wild following their release. (nine fledglings from 32 eggs is a nest success rate of 28.1%. Natural rate for wild helmeted honeyeaters has been recorded as 31.6% (Franklin *et al.*, 1995)). Three of these are known to have attained sexual maturity, but none has established a breeding territory at the release site.

Establishment

A resident social group of helmeted honeyeaters has not yet formed at the re-introduction site. However, good survivorship of both adult and juvenile released birds, successful breeding by a number of pairs following release and the survivorship of consequent offspring are successes of the re-introduction trials undertaken to date.

The future

The recovery plan for the helmeted honeyeater has the goal of re-establishing the bird more widely within its former distributional range (Menkhorst *et al.*, 1999). Re-introduction trials will continue with the aim of refining techniques, particularly to achieve the establishment of resident communities of birds. With a view to improving the bird's conservation status, a second release site is currently being set up. This location is in a different catchment from that inhabited by the Yellingbo population. It is expected that helmeted honeyeaters will be released there during 2000/01.

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A review of noisy-scrub bird re-introductions, Australia

Alan Danks

Introduction

The possibility of re-introducing the noisy-scrub bird *Atrichornis clamosus* to other parts of its former range was discussed soon after this long-lost species was rediscovered at Two Peoples Bay in 1961. The small group of survivors was largely confined to the Mt Gardner area but full recovery of the species demanded both larger numbers than were likely to be supported in the limited habitat and a more widespread population to reduce the potentially disastrous effect of wildfire. With poor dispersal powers the scrub-bird would depend on human assisted translocations to colonize other areas.

To provide stock for a re-introduction program a captive colony of noisy scrub-birds was established in 1975 using nestlings taken from Two Peoples Bay (Davies *et al.*, 1982). Breeding scrub-birds in captivity however proved difficult and the program was abandoned in 1981. Fortunately, by this time the number of noisy scrub-birds in Two Peoples Bay Nature Reserve had grown and it was considered acceptable to take small numbers from the wild population for re-introduction (Danks, 1997).

Pilot translocation project

A trial translocation project began in June 1983. The first release site was on the northwest flanks of Mt Manypeaks, 15 km east of Mt Gardner, an area that was almost certainly within the species former range. A major aim of the pilot project was to develop routinely applicable methods for capturing, holding and transporting noisy-scrub birds. These birds however, are elusive, cryptic inhabitants of impenetrable scrub. They are only rarely seen, and only one adult had ever been captured. The development of reliable capture techniques was the most difficult and time-consuming part of the pilot project.

Song playback in conjunction with a specially designed and actively operated mist net eventually proved effective for territorial males. Females could be trapped at the nest. Elliot mammal traps were also used for catching males and females of various ages where the density of individuals was high (Danks, 1997). From June to August, males are actively defending territories and the females are nesting. Outside the breeding season, capture was difficult and unreliable.

During the pilot project, holding aviaries were built at Two Peoples Bay so that captured birds could be held and maintained in temporary captivity. The establishment of singing territories by the males provided an indication of persistence at the release site. Counts of singing males (the population index) was a reliable way to follow population trends at Two Peoples Bay and this was used to monitor the effects of removing birds from the parent population.

Female scrub–birds proved to be more difficult to capture than the males and only six females accompanied ten males in 1983. As the males were captured early in the season, they were released first—one or two birds at selected sites. A female was then released where a male was singing.

The persistence of four territorial males in the release area through 1984 was very encouraging. In 1985 a further seven males and seven females were added, although there was no evidence of breeding or longer term viability at this stage. Indications that breeding had occurred in the new colony were not apparent until 1987 when the number of singing males exceeded the number released (See Table 1).

Re-introduction program

Based on the apparent early success of the Mt Manypeaks re-introduction, and using the same techniques, scrub-birds were translocated to two other areas on the south coast to the west of Albany: Nuyts Wilderness in Walpole-Nornalup National Park and Quarram Nature Reserve (Danks *et al.*, 1996 & Danks, 1997). By 1990 these three sites had received a total of 88 (48:40) noisy-scrub birds. Only the Mt Manypeaks translocation was successful and the birds failed to persist at the other sites (Table 1).

During this time, the holding aviaries were expanded, insect breeding and storage facilities were developed and it became routine to hold birds for periods of up to three weeks when necessary. They were transported for 1–3 hours in vehicles followed by 1–2 hours in backpacks to reach the more remote release sites. No attempt was made to acclimatize the birds to the area before release. Capture equipment and techniques became more dependable with experience. Nevertheless, considerable effort was required over two seasons to capture 25–30 individuals.

Generally, the birds could only be captured one at a time, with sometimes days or weeks between captures. When four to six birds had accumulated in the holding aviaries at Two Peoples Bay they were transported to the release site.

New strategy

From 1990, a different strategy was used based on the need to reduce the potential for losses (of females in particular) at release sites that proved to be incapable of supporting scrub–birds. The initial release groups were reduced to 4–8 males often with one, sometimes without any females. The survival and persistence of some or all of these males until the next breeding season provided verification of the habitat before any females were added.

At the same time, release site selection procedures were refined to include an assessment of food supplies (leaf litter invertebrate abundance) as well as vegetation structure and composition. Radio tracking was used successfully at two release sites providing information on post–release behaviour and interactions between birds. Monitoring of the source population showed that males were being replaced and the Mt Gardner population continued to increase despite the regular, almost annual removals over the 12 years from 1983 to 1994 (Danks, 1997).

In this second series of translocations between 1990 and 1994, a total 37 (23:14) scrub-birds were released at four sites-all within 50 km of Two Peoples Bay. This was a more successful strategy and scrub–birds persisted at three of the four release areas.

The release of noisy-scrub birds on Bald Island between 1992 and 1994 was really an introduction rather than a re-introduction, as scrub-birds were not previously known from the island. There were several conservation advantages to be gained from establishing an island population however, including isolation from mainland predators and diseases and a reduced fire frequency in addition to adding to the overall population size.

Table 1. Summary of noisy scrub–bird translocations from Two Peoples Bay between 1983 and 1999

RELEASE SITE	DISTANCE AND DIRECTION	RELEASE YEARS	TOTAL MALES RELEASED	TOTAL FEMALES RELEASED	SINGING MALES AFTER 2 YEARS	YEAR SINGING MALES EXCEEDED NUMBER RELEASED
Mt. Many Peaks	15 km. (East)	1983, 1985	17	14	12	1987
Nuyts	150 km. (West)	1986, 1987	16	15	2	-
Quarram	120 km. (West)	1989, 1990	15	11	0	-
Mt. Taylor	15 km. (West)	1990, 1992, 1993	6	6	5	1993
Mermaid	20 km. (East)	1992, 1993	4	4	1	1999
Bald Island	25 km. (East)	1992, 1993, 1994	8	4	6	1997
Stony Hill	25 km. (West)	1994	5	0	0	-
Darling Range						
1. Samson Rd.	310 km. (NW)	1997, 1998	6	4	3	-
2. Upper Harvey	300 km. (NW)	1997, 1998, 1999	23	4	5	-
3. Falls Brook	300 km. (NW)	1998	5	0	0	-
TOTAL 1983–1999			105	62	34	

The Darling Range re-introduction

Between 1994 and 1996, searches for other potential release sites in the Albany area and inland to the west failed to locate any area with sufficient habitat to allow the development of a large population. The Albany area was only one of three 19th century scrub-bird populations. The type locality for instance, was in the Darling Range near Mt William, 100 km south of Perth and 300 km northwest of Two Peoples Bay. Surveys in 1996 and 1997 confirmed the presence of habitat apparently suitable for noisy-scrub birds along the streams, which rise in the high rainfall uplands of this part of the Darling Range.

The re-introduction of noisy-scrub birds into the Darling Range began in 1997 when three groups of five males were released at three separate areas of similar habitat along streams. The persistence of singing males at two of these sites encouraged the release of females and more males in 1998 and 1999. At one site, in the Upper Harvey River area, radio tracking showed that predation was contributing to the apparent losses of released birds. Larger numbers of mostly male scrub-birds have been released at this site to compensate. In late 1999, eight males were regularly defending territories in two of the three areas in the Darling Range.

Technology transfer

The methods used to capture territorial male scrub–birds are applicable to other ground dwelling, territorial birds inhabiting dense vegetation. Consequently, in 1999 the first translocation of western bristlebird *Dasyornis longirostris* from Two Peoples Bay made use of this technique as well as the facilities for holding, feeding and transporting birds that had been developed and refined over many years for the noisy-scrub bird re-introduction program.

Fire management

Noisy scrub–birds are sensitive to fire. Being semi-flightless, scrub-birds depend on the cover of dense understory shrubs and sedges. Fire removes this understory and it may take several years to grow back. Likewise the leaf litter invertebrate fauna may require even longer to develop sufficient diversity and abundance to support noisy-scrub birds. Effective fire control is a challenge in fire-prone vegetation communities of the southwest of Western Australia but it is crucial in the early years of development when the new population is small and vulnerable. The importance of this was demonstrated at the Mt Taylor re-introduction site when a wildfire in 1995 burnt out 75% of the small but apparently thriving scrub–bird colony. The few survivors then disappeared after a very dry summer in 1996.

Achievements

Over the 17 years since 1983, a total of 167 noisy scrub–birds have been translocated from Two Peoples Bay Nature Reserve to 10 sites. At four of these sites the birds established breeding populations (Table 1). Unfortunately, one of these was lost to wildfire. At two of the three Darling Range sites, the birds appear to be persisting and there is every chance they will also breed.

The primary achievement of the re-introduction program has been its contribution to population growth. The Mt. Manypeaks area in particular has been very productive and now supports the largest noisy-scrub bird population. In 1999 total noisy-scrub bird numbers were estimated at 1470 individuals. Approximately 1000 (68%) of these were in populations which were begun by translocations.

Translocations have also resulted in an increase in the number of populations. In 1983, there were two sub-populations, both within Two Peoples Bay Nature Reserve. By 1996, eight sub-populations could be distinguished in the Albany area occupying about 40 km of coastal and near coastal country. Six of these had been created by translocation (Danks, 1997). Although several of these formerly separate sub-populations have now coalesced due to population growth, the greater spread of scrub-birds across the landscape has significantly reduced the species' vulnerability to wildfire.
The population growth since 1993 and greater geographic spread were largely responsible for the noisy-scrub bird being moved from the IUCN category Endangered, to the lower threat category Vulnerable in 1998.

Support

The Western Australian Department of Conservation and Land Management is responsible for conservation of the noisy-scrub bird and the continuity of resources, effort and professional staff has been one of the recovery program's strengths. Support has also come from the Commonwealth Government (Environment Australia), local government and non-government organizations like Birds Australia as well as tertiary education institutions (Murdoch University, University of WA) and the New Zealand Department of Conservation.

In recent years, the mining company Alcoa Australia has also contributed financially to the Darling Range re-introduction. The Noisy-scrub Bird Recovery Team and South Coast Threatened Birds Recovery Team have provided additional support for the translocation program and an important forum for discussion and decision making.

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The re-introduction of malleefowl to Shark Bay, Western Australia

Colleen Sims

Introduction

The malleefowl *Leipoa ocellata* is one of three megapode species present in Australia. It is the only species of the 22 members of the Megapodiidae, predominantly restricted to semi-arid and arid environments, in southern Australia.

It has suffered considerable range reductions throughout much of Australia since European settlement and is classified as endangered by the Commonwealth Endangered Species Act, 1992, and vulnerable nationally (ANZECC, 1995). In Western Australia, much of its core habitat has been cleared for farming, but range reductions have been less marked, and it is listed as vulnerable (Schedule 1) on the Threatened and Priority Fauna and Flora List (CALM, 1999). The principle threats to its survival is recognised as loss and fragmentation of suitable habitat, and predation by the red fox *Vulpes vulpes* (Benshemesh, 1999; Brickhill, 1987; Jones, 1989; Priddel & Wheeler, 1997).

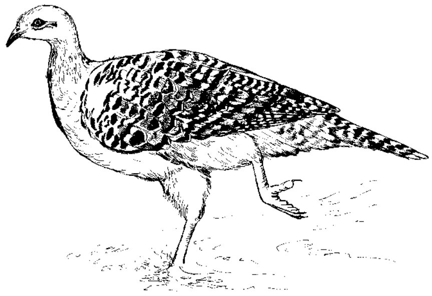
There are a number of records of malleefowl on Peron Peninsula prior to 1950 (Benshemesh, 1999), but only occasional sightings on the northern peninsula in recent memory. The species maintains a presence in the south west of Shark Bay. The Peron Peninsula in Shark Bay, is an area of approximately 105,000 ha of semi-arid habitat that was formerly used for extensive sheep grazing for over 100 years. In 1990, the area was acquired by the Department of Conservation and Land Management (CALM), and an extensive program of de-stocking carried out. In 1994 a feral barrier fence was erected across a 3.6 km isthmus and feral control extended to include eradication of sheep and foxes, and reduction of the feral goat population to an estimated 500-1000. The population of the European rabbit *Oryctolygous cuniculus* fluctuates seasonally, but remains relatively stable over the long term, and the feral cat population has been reduced to an estimated 20% of former numbers by trapping and '1080' (monofluoroacetate) poison baiting.

The feral control program has been instigated as a precursor to re-introductions of a large number of native faunal species, which have become locally extinct since European settlement. The malleefowl is one of the few species that has managed to persist elsewhere on mainland Australia in the face of feral predation and habitat destruction, and was therefore selected as the first species to be re-introduced to this area.

Egg Collection and incubation
In late spring of 1996 and 1997 active wild, malleefowl mounds in south west Western Australia were located and eggs were collected from these mounds in November/December of each year. These were transported to Shark Bay to be artificially incubated and raised, before releasing the chicks onto Francois Peron National Park (the most northern 40,000 ha of the Peron Peninsula) the following winter/spring.

A maximum of half of the total clutch, were collected from each mound. The mounds were selected on the basis of recent activity and excavated only once to remove the eggs. Age and development of eggs were estimated by 'candling' (Sims, in Prep). Only those eggs with advanced embryos were collected for incubation. A total of 105 eggs were collected, from which 90 chicks hatched.

Chick rearing
After chicks hatched, they were kept in brooder pens until they reached a minimum of 140g in weight. A microchip transponder was then implanted into the breast muscle for permanent identification. Chicks were then transferred to 10m x 10m aviaries in groups of 8-12. The enclosures were made of chicken mesh walls to ~1.8m high and covered on the inside by 90% shade cloth. The roof was made from nylon bird netting, arched from the top of the walls to ~5 m high. A rabbit-wire mesh was used to provide an external skirt of 1m width, buried at the base of the walls to prevent predators digging in. Natural vegetation of predominantly *Acacia* shrubs was retained inside for cover and roosting.


Malleefowl *Leipoa ocellata*

Chicks were fed on a diet consisting of a mixture of small and large grains, turkey starter, mealworms and vegetable greens and fungi (Sims, in prep). The size of food items was increased as birds grew.

Once in the aviaries, chicks were left undisturbed with minimal handling, unless required for health reasons. Feeding occurred once a day. Malleefowl, like most galliformes are very flighty, reacting to stimulus by exploding skyward like a pheasant. They can be partly tamed by hand feeding of favoured food (e.g. mealworms), but this is temporary and any unusual disturbance by husbandry staff (e.g.. attempts to capture an individual for examination) created a severely heightened flight response to human presence for many weeks, before they become habituated again.

Chicks were grown to between 6–12 months of age and 800–1690 g body weight, before release. Of 90 chicks hatched, a total of 67 were released over the 2 years.

Radio-tagging
The Biotrack radio-transmitter had a ~6 month life, and range of 3 km with a mortality signal added. Radio-tags were attached using a necklace arrangement, which has been used successfully in many galliformes in the northern hemisphere. The tags did not have a true breakaway mechanism, but a weak point was created by an acrylic glue, which broke down after an unpredictable exposure time (1-6 months). Initial plans to recapture and remove tags prior to battery exhaustion proved impossible in the dense *Acacia* scrub habitat that most birds selected. Given the benign nature of the necklace and its inherent weak point, we were confident that all tags would eventually be lost without detriment to the birds.

Initially, birds were radio-tagged with the necklace and returned to the aviary for a minimum of one month to observe for any changes in behavior prior to release. Malleefowl appeared to accept the necklace well with no detectable change in behavior or flying ability. No signs of injury or chafing were detectable, compared to other tag designs previously tried on malleefowl (Benshemesh, 1992).

Capture and processing.
Once the benign nature of the radio-tags was established, all subsequent birds were caught only once, and all processing (banding, measurements, weighing) done on the day prior to release. Birds were kept inside a cotton handling bag and carry in a quiet, cool place before being released before sunrise the following day. Release sites were selected for maximum vegetation cover (Priddel & Wheeler, 1997).

Birds were caught prior to release by hand netting. Catching large numbers of birds at once resulted in considerable stress to some birds and mortalities occurred. Night netting with spotlights was not effective as birds were startled more easily when disturbed at night, and all birds in the aviary took flight together. A manual drop net over a feeding station would be recommended for future programs as a preferred capture technique.

Malleefowl are prone to developing hyperthermia when stressed, and handling time must be kept to a minimum. Holding of birds must also be planned carefully to prevent individuals crowding together and exacerbating the problem. After some early problems, birds were caged individually and kept in air-conditioned environment with good air circulation to minimize this.

Monitoring

Twenty-four out of a total 67 released malleefowl, were radio-tagged and tracked for a few days to 6 months post-release. Tracking was performed daily for the first few weeks, then reduced to 2-3 times per week for 1-3 months, then once every two weeks. Permanent omni-directional antenna on 30 m towers were used, in combination with hand-held Yagi antenna on a 6 m mobile aerial. Not every bird was located daily, but all signals received, were monitored for the mortality. Four of these 24 radio-tagged birds died during monitoring (Table 1.). This limited sample suggests a survival rate of over 80% to six months post-release. The longer a signal went undetected, the more difficult it was to relocate, as birds were sometimes found to have moved as much as 10-15 km in a day and changed direction of movement from one day to the next. Aerial tracking was used to relocate signals on several occasions. During the day the signals often fluctuated significantly in strength (probably due to foraging activity), and tracking at night was found to produce better results, as birds were generally roosting several metres above the ground in trees.

Long-term monitoring has consisted of opportunistic sightings of birds and their footprints by park and feral control staff, who are patrolling sand tracks across the area on a daily basis. Public sightings are also recorded. There are currently plans to set up monitoring grids in 2000, to search for birds and their nesting mounds, as the oldest individuals will now be 3 ½ years old and approaching breeding age.

Discussion

Assessment of success of program: Malleefowl are a long-lived species (30+ years), with late onset breeding (4-6 years) and low recruitment (<1% survival of chicks in the wild). Despite this, it was possible to quickly, and economically produce large numbers of young birds for a re-introduction program by collecting eggs from the wild. This method prevented the need to maintain a large breeding population for long periods in captivity, and removing only half the eggs was unlikely to significantly impact the reproductive capacity of the wild birds. In fact it could be used to improve the recruitment rate of a recovering wild population by releasing older, less vulnerable chicks (Priddel & Wheeler, 1996) back into the same population, after the age of highest mortality (80% in first two weeks).

Table 1. Mortality of released malleefowl

ID	WEIGHT AT RELEASE (g)	AGE AT RELEASE (DAYS)	TIME OF DEATH POST-RELEASE (DAYS)	CAUSE OF DEATH
F19	1400	261	67	Aerial predation
F28	1000	285	420	Trapped (1800 g)
F88	980	233	8	Stress associated disease
F7	1280	306	~ 11	Starvation

Because of the highly temperamental nature of this species it has been found that all unnecessary handling and interference should be avoided to prevent injuries and stress-related mortality of chicks. Enclosure design, with high, soft roofing was probably an important factor in reducing injury when birds were disturbed. There is also some evidence, comparing weight for age between handled and unhandled birds, both within this program and between this and other programs where chicks were captured regularly for weighing, that a significant decrease in growth rate results from the stress brought about by even a single capture event.

The importance of stress for the overall health, growth, reproduction and ultimate survival of all wild and captive species can significantly affect the success of any reintroduction program, and is a factor that I believe is often overlooked or seriously under-rated in conservation biology. The impact of husbandry and handling as well as marking and monitoring practices on the stress level of all species must be critically evaluated at all times (Cox & Afton, 1998). The addition of any extra stressors on top of natural ones (e.g. gestation, lactation or male competition) and those already inherent in a translocation, can make the difference between success or failure. Priddel and Wheeler (1990), highlighted the effect of food availability on predation, but the cause and results of stress may be more subtle and not always recognized, when the ultimate cause of mortality or decreased reproduction may be identified as something else.

Changes in handling and holding procedures reduced mortality in later releases during this re-introduction, and modified capture techniques would further reduce the risks and stresses during translocation. There is little doubt from survivorship of radio-tagged birds and previous studies by Priddel and Wheeler, that greater age and body weights at release, combined with an absence of the principle ground predator in the fox, and dense, low vegetation contributed greatly to the early success of this program. Effective monitoring of radio-tagged birds, which move such distances must take into account the need for aerial tracking and could have been improved by routine night-time monitoring. A successful capture technique for birds in dense scrub vegetation would also allow replacement of tags for longer term monitoring of survival.

We are yet to determine the long-term success of this re-introduction program. Another 5-10 years will show if released birds have successfully produced reproductive offspring. This will require a longitudinal banding study of the population in addition to the long-term monitoring of nesting mounds. The ultimate goal is for the population to expand to fill the available habitat, and even to spread south, mixing with the remnant, extant populations south of the re-introduction area.

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Several papers detailing this re-introduction program are currently in preparation by the author and will be submitted to Wildlife Research for publication in 2000/2001.

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Re-introducing shore plover to "Mainland" New Zealand

Shaun O'Connor

Introduction

New Zealand shore plover *Thinornis novaeseelandiae* is a rare shorebird endemic to New Zealand. The species was formerly widespread around the New Zealand coastline, however, disappeared from the mainland in the 1800's following the introduction of mammalian predators, particularly Norway rats *Rattus norvegicus* and cats *Felis catus*. Shore plover have been confined to two wild populations on islands free of introduced predators in the Chatham Islands (800 km east of New Zealand) for the past 100 years. South East Island (Rangatira) holds a self-sustaining population of 130 birds, which has reached carrying capacity within the Island's available habitat. A second small population of 21 birds was discovered on a small reef in the Chathams in 1999.

Because only one robust population of shore plover remains, it is highly vulnerable to extinction. Threats such as accidental introduction of mammalian predators or periodic catastrophic events such as fire or disease could have a severe impact on the species and potentially drive it to extinction. Shore plover are ranked as Endangered in the IUCN Red List Categories (IUCN, 1994).

The endangered status of shore plover has spurred conservation efforts and a recovery programme for the species administered by the Department of Conservation and coordinated by the Shore Plover Recovery Group.

The draft recovery plan has a 10-year timeframe (2000–2010) with a 10 year goal to maintain and/or establish wild shore plover at a total of five or more locations with a combined population of 250 or more birds. The plan has a five-year goal to protect current self-sustaining populations of shore plover on the Chatham Islands and establish at least one new population in the wild in New Zealand.

The Shore Plover Recovery Program

Attempts to reduce the risk of extinction by establishing new populations of shore plover began in the 1970s. Three transfers of birds from South East Island to predator free Mangere Island in the Chathams were undertaken between 1970 and 1973 (Aikman, 1995). Both adults and juveniles were translocated. Unfortunately these transfers failed primarily because transferred birds exhibited very strong natal and/ or territorial site fidelity by flying back to South East Island soon after release.

A second re-introduction technique of captive breeding/release was initiated from the 1980s. Eggs were collected from South East Island and transferred to captivity for artificial incubation and hand rearing to form a captive population. The aim was to establish a self-sustaining captive population to produce surplus birds for release without having to crop the South East Island population regularly. It was also thought that birds reared in captivity may be less site faithful (i.e. to captive institutions) than wild caught birds are to their natal site, and therefore more likely to stay at release sites. Experimental releases would be trialed to attempt to circumvent the site fidelity trait and identify the optimum "release design" to enhance post release survival, residency and ultimately breeding at re-introduction sites.

Initially, fresh eggs were transferred in the 1981/82 and 1982/83 seasons to the Mt. Bruce National Wildlife Centre and Otorohanga Zoological Park in New Zealand (Anon, 2000). These early attempts produced poor hatching success and long term survival. Further egg transfers were undertaken in the 1990/91, 1993/94 and 1995/96 seasons. Eggs were transferred from mid-term incubation and hand-rearing techniques were fine-tuned with each transfer. By the 1995/96 transfer, techniques had been refined sufficiently to produce success rates of 100% at hatching and 95% at fledging. During this period two inter-dependant captive populations were established at the Mt Bruce National Wildlife Centre and at Peacock Springs (Isaac Wildlife Trust) in Christchurch.

By 1994, the captive population was producing enough juveniles annually to initiate the re-introduction program. The Shore Plover Recovery Group drafted a list of potential release sites for shore plover based on the known suitable habitat of South East Island and the absence of introduced mammalian predators. Following a detailed habitat assessment by Davis (1994), Motuora Island in the Hauraki Gulf was selected as most suitable for re-introduction of shore plover.

Motuora re-introduction program

Six releases of captive-bred shore plover were carried out on Motuora Island between 1994–1999 and closely monitored. Samples of release birds were small (4–18), taking only the surplus of progeny produced annually by the captive breeding population. It was therefore thought necessary to run this release program for five years to fully test and analyse the technique at this site with a reasonable dataset in terms of survival, residency and breeding of released birds. A research by management approach was adopted and the "design" of releases modified annually to overcome problems encountered and to attempt to determine the optimum release design that would maximise the likelihood of establishing a population of shore plover on the island. Design variables manipulated included: the time of year of release, length of pre-release holding period in aviaries (conditioning to site) on the island, age classes of released birds, gender ratio and rearing class of released birds. Unfortunately the gender of juvenile shore plover can not be established from plumage until the birds are at least eight months old, so releases involving juveniles could not be manipulated by gender ratio (possible via blood/ DNA from 1999).

All birds released were individually colour banded and at least 75% of released birds were fitted with transmitters for monitoring purposes each year.

Performance measures were set for each release. Generally they included determining the fate of 75% of released birds one month after release, ensuring all birds were colour banded and at least 75% transmitted. Changes to subsequent release designs were based on survival and residency seven months after release.

A total of 75 birds have been released on Motuora since 1994 (Table 1.). The monitoring results have revealed that captive reared birds are well adapted to foraging in the wild with active feeding and healthy weights being achieved.

Results of the first three releases were evaluated by the Shore Plover Recovery Group based on residency at September 1st 1996. Ten birds (27% of released birds) were resident. When analysed by different classes of birds released (identified in Table 1) we found that 60% of residents were birds released as juveniles (2/3 month and 10 month combined), although they make up 75% of the sample size of the 2nd and 3rd releases. Seventy percent of the residents were male and 44% of residents were hand-reared birds compared to 22% parent reared birds. Analysis by holding period was inconclusive. Interestingly, following the September evaluation, regular weekly monitoring found a more fluid pattern of residency and a decline in numbers from September.

Aikman (1999) presents results of the first four releases from an evaluation of residency at September 1st 1997. She found that: the fate of 60% of the released birds is known. From this group dispersal (53% of birds) had been the major cause of loss from Motuora. Predation (probably by morepork *Ninox novaeseelandiae*, a native owl) accounted for 13% of birds of known fate. In terms of residency Aikman (1999) found that 74% of birds disappeared from the island during the first month after release. Of the 14 birds that remained on the island for at least one month, eight (57%) were present on the 1st September 1997, 6–23 months after their respective releases (Aikman, 1999). Generally, once birds had stayed for the first month following a release, residency was more stable through to September.

When analysing by different bird classes Aikman (1999) found that: more adults than juveniles had unknown fates (71% of adults c.f. 28% of juveniles), however a much higher proportion of adults remained resident for longer than one month (57% of 14 adults c. f. 26% of 39 juveniles). She found that there was no significant difference between the fates of hand reared and parent reared birds.

Monitoring of residency showed a link between the onset of the breeding season (September–October) and shore plover dispersal from Motuora. Aggressive behaviour within the shore plover flock as birds paired and became territorial could encourage dispersal of subordinate birds from the island. However, there is considerable unoccupied habitat/territory on the island for subordinate birds to move into rather than disperse and at the onset of each season the shore plover population was relatively small <10 birds. It is more likely that birds were being encouraged to leave by another factor—seven predations of shore plover by morepork had been recorded over this same timeframe following releases. The limited life expectancy (30 days) of batteries in the small transmitters and problems with transmitters falling off birds prematurely in the early releases has meant that the exact fate of 36% (27 birds) of birds released can not be determined. The incidence of direct predation could therefore be higher. A significant proportion of released birds (18% in June 1998) had been sighted in adjacent mainland coastal areas in the Auckland region following dispersal from Motuora.

This dispersal pattern suggests that not only were morepork directly predating shore plover but also that active hunting pressure by morepork is likely to be scaring shore plover from the island. Morepork are a spring breeder laying eggs from September–February (Heather & Robertson, 1996) so there is a direct link with the dispersal timeframe.

The morepork–shore plover relationship on Motuora is likely to be unique. The island is highly modified (exotic grassland with a fringe of native and exotic trees) and has a limited range of traditional prey for morepork. The island's insect, reptile and small bird fauna is depauprate and the island is rodent free. The requirement for prey would obviously increase when morepork are putting on breeding condition and feeding young (September–February). Shore plover presumably co-existed with morepork before shore plover became extinct in mainland New Zealand, however, the natural prey populations would have been much greater and more diverse than that remaining on Motuora today. In another twist, morepork are not present in the Chatham Islands and there is no record of them ever being present, so it is likely that shore plover derived from South East Island are naïve to this nocturnal predator.

The morepork issue came to a crux in the 1998/99 releases. Eighteen birds had been released and there was clear evidence of further morepork predation (three birds preyed on within five days of release) and harassment (four birds dispersed within two days of release and 11 birds dispersed within two months of release). With this further evidence, the program on Motuora was

Table 1: Shore plover released on Motuora Island

YEAR	MONTH	BIRDS RELEASED	HOLDING PERIOD	AGE CLASS	GENDER	REARING CLASS
1994	September	5 (of 8)	1 month	All juvenile (<10 months)	3m/2f (2m/1f)	All parent
1995	September	15	1 month	All juvenile (<10 months)	8m/6f/8unk	12 parent 3 hand
1996	February	16	adults-2 weeks Juvenile-4 days	8 adults (1-3 years) 8 juveniles (2/3 months)	4m/4f/8unk	10 parent 6 hand
1997	February	17	adult-2 days juvenile-5 days	6 adults 11 juveniles (2/3 months)	3m/3f/11unk	11 parent 6 hand
1997/98	Dec 1997 to Feb 1998	18 (of 36)	1 month	30 days approx.	1m/2f/15unk	All hand
1999	June 1999	4	2x2 weeks 2x6 months	2x3 5 years 1x7.5 years 1 juvenile	3m/1f	1 parent 3 hand

NB: in 1994 only 5 of 8 birds were released as 3 birds were preyed on while in the holding aviary by a harrier *Circus approximans* which had pulled birds through the aviary mesh. In 1997/98 large scale releases were postponed mid way through the release program so only 18 of the scheduled 36 birds were released.

postponed midway through the 1998/99 releases. The Shore Plover Recovery Group recommended that either the estimated five pairs of morepork be removed from the island (e.g. by translocation) to allow for a further 3-5 years of releases on Motuora in the absence of morepork pressure or, if this was not possible, a second release site be chosen that has suitable habitat and lacks both introduced predators and morepork.

Local Iwi (indigenous Maori tribes) were consulted with regard to the potential capture and transfer of morepork to a mainland site. Iwi were unanimously opposed to manipulation of the morepork as morepork hold very strong spiritual values within the Iwi culture as kaitiaki (ancestral caretakers). Department of Conservation's senior managers made the decision not to manipulate morepork out of respect for these strong cultural values (recovery groups can make recommendations only). The Shore Plover Recovery Group therefore identified and ranked three alternative sites for assessment and choose the highest-ranking site as it fulfilled the necessary criteria and did not have a morepork population. The island has sand shoreline on one point but is dominated by hard rock shoreline and extensive wave platforms at low tide.

Re-introduction program at second release site

NB: As the 2nd release site is privately owned and the owners do not wish to attract publicity, the name of the site has not been included here.

Fifteen birds were released as a trial at the 2nd release site in August 1998. All birds were hand reared, juveniles <10 months old and of unknown sex at release. All birds were fitted with transmitters. Birds were held for 10 days in pre-release holding aviaries. Post-release survival and residency has been high at the 2nd site. One bird dispersed to the neighbouring mainland coast in the first month following release and 14 birds were still resident three months after release. Territorial and pairing behaviour were recorded over the 1998/99 season, however no active breeding behaviour was recorded. Residency was high at 60% after 12 months and 53% after 18 months following release.

A second release, of 10 juveniles (6/7 months old) of unknown sex, was undertaken in July 1999. Most were parent reared. The birds were only held for three days in the pre-release holding aviary. October 1999 territorial aggression levels rose noticeably, particularly amongst the first release birds. Three birds dispersed to the mainland between September 1999–February 2000. Five pairs subsequently formed over the season and successfully bred, fledging four chicks over 1999/00. At the end of March 2000, 19 birds were resident at this site (14 adults and five island-reared juveniles). A further release of up to 15 juveniles is planned for May 2000.

In a parallel with the Motuora program, dispersing birds from the 2nd re-introduction site have been seen several hundred kilometres from the release site, apparently surviving on the mainland coast in the presence of a suite of introduced predators.

Motuora postscript

Over the 1998/99 season Motuora recorded the first successful breeding of wild shore plover on mainland New Zealand in the 20th century. Two pairs laid two fertile clutches of eggs over November/December 1998 and one clutch hatched on Christmas day. The 2nd pair's clutch failed when the male disappeared at night during incubation. Two chicks fledged from the successful clutch however both died-one from a leg injury and the other from harrier predation.

A small release of four birds was undertaken in 1999 to provide pairing opportunities for the single female still resident from the former second pair.

In the 1999/ 2000 season the same first pair successfully bred again and fledged one chick on Motuora. This chick dispersed to a neighbouring island where it still remains (at the time of writing) with a male released in 1998. The successful pair is still resident on Motuora; they have obviously learnt to survive in the presence of morepork. Both were parent reared and released as juveniles.

What have we learnt from the re-introduction programs to date:

⇒ *Assumptions about suitable habitat based on the last refuge of the species.*

In selecting the first re-introduction site we had assessed potential release sites on the optimum habitat found on South East Island. South East Island has predominantly rock wave platform. One of the reasons Motuora ranked highly as the first release site is that it has similar wave platforms on its eastern side. However, it has more diversity of habitat with sandy beaches and a mix of sand and wave platform on its western side. Released birds on Motuora displayed a preference for the sandy areas on Motuora. In addition, birds that dispersed from both Motuora and the 2nd release site showed a preference for sandy coastline with sightings at river mouths on sandy beaches and in estuarine habitat. This assumption has parallels with other endangered species programs (e.g. takahe *Notornis mantelli*) where there is a temptation to assume that the habitat that the species still occupies is its optimum habitat.

⇒ *The re-introduction program has shown us that fully-grown shore plover are able to cope to some extent with the full suite of introduced predators on the mainland of New Zealand.*

Some dispersed birds have been able to survive for up to two years (from incidental sightings) in the presence of two rat species (*Rattus norvegicus* and *Rattus rattus*), cats, stoats *Mustela erminea*, weasels *Mustela nivalis*, ferrets *Mustela putorius*, dogs and aerial predators. This has challenged previous assumptions about the South East Island population. It has been generally thought that because the South East population is at carrying capacity, juveniles disperse to neighbouring Pitt Island and are preyed on by cats. Shore plover are occasionally sighted on Pitt Island, however, given the results of the re-introduction program you would expect to see more adult shore plover surviving/ residing on Pitt. This finding also indicates that shore plover indeed do have very strong natal site fidelity. If this was not the case we would expect to see more fully grown dispersed birds on Pitt Island in the Chathams as overflow (not in a breeding capacity on Pitt) from South East Island.

⇒ *The re-introduction programs have taught us that no matter how much effort is put into manipulating the release design by trialing distinct classes of birds, there may still be an unpredicted biological factor in play that jeopardises the ability to objectively analyse results and identify optimum release designs.*

The background influence of morepork predation pressure/ harassment (encouraging dispersal of released birds) jeopardised the ability to analyse results and identify the optimum release design. Conversely the recent success of the two releases of shore plover at the 2nd site has been achieved under a number of variations of "bird classes". The species may be easier to re-introduce than indicated by the Motuora results.

⇒ *The importance of thorough site assessment is critical to any re-introduction programme. In assessing Motuora Island for shore plover the Shore Plover Recovery Group failed to anticipate the potential effect of morepork as an aerial predator.*

It may still be possible to establish shore plover on Motuora (as evident from the current resident breeding pair), however a long timeframe and high losses would be expected.

⇒ *Robust monitoring tools are essential for re-introduction programs.*

The failure of transmitter attachment techniques early in the Motuora program meant that the fate of a significant percentage of birds could not be determined and the likely influence of morepork took longer to identify. Even now we can not be certain that morepork are the prime reason for failure to establish a sizeable population on the island, however it seems likely.

⇒ *Performance measures for releases need to be realistic, particularly during the establishment phase.*

Impediments (especially unforeseen ones) are likely to have a greater impact on a smaller, establishing population and while newly released birds are initially naïve to their new home. These impediments may be acceptable in an established population (e.g. predation by natural predators) but are likely to have a greater impact on a newly establishing population.

⇒ *Recovery Groups for threatened species are by nature (at least in New Zealand) groups of technical expertise which are mandated to recommend best technical advice to managers undertaking re-introduction programs.*

Recovery groups do not make management decisions, particularly in contentious situations. The Motuora program saw a conflict in interests and values between species recovery/ecological restoration and cultural values when the impact of morepork on shore plover emerged. The simple answer for the recovery group was to remove morepork during the re-introduction/establishment phase. However from consultation with indigenous iwi it became clear that any manipulation of morepork was unthinkable as morepork held very high spiritual values for these people. This was a difficult time, but a valuable lesson, for the recovery group. Each perspective on values is plausible in its own right and a pragmatic approach is needed to find a solution.

Challenges ahead

1. Restoration of shore plover on mainland New Zealand (not just inshore predator free islands) in association with sustainable habitat management, particularly predator control. This concept is being developed in pioneering “mainland island” programmes with forest birds in New Zealand.
2. Restoration opportunities within the Chathams archipelago. Ideally the formation of a number of island populations as a larger meta-population would be useful to spread the risk of threats establishing in any one population and to allow genetic flow between these populations. This was presumably the natural pre-human situation.
3. Ultimately, achieving the recovery goals and reducing the status from endangered to vulnerable within the 10 year recovery plan timeframe.

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Re-introductions of New Zealand robins: a key component of ecological restoration

Doug Armstrong

Background

The New Zealand robin “toutouwai” *Petroica australis* is a small (26-32g) ground-feeding insectivorous forest passerine. It is one of three members of the Australasian robin family (Eopsaltriidae) endemic to New Zealand, the others being the black robin *Petroica traversi* and tomtit *Petroica macrocephala*.

Black robins are the most famous story in re-introduction biology, the species having been recovered from five birds translocated from Little Mangere Island to Mangere Island in 1976. This story has featured in numerous articles and films, and is covered comprehensively in Butler and Merton's (1992) book "The black robin: saving the world's most endangered bird". The species' fame is such that most New Zealanders immediately think "black robin" at the mention of the word "robin". However, fewer people realize that black robins are confined to the Chatham Islands 700 km to the east of the main islands of New Zealand, and are have a more common non-black (I like to say "non-mutant") relative.

New Zealand robins are still found on all three of New Zealand's main islands, and are considered to have three races (North Island robin *Petroica australis longipes*, South Island *Petroica australis australis* and Stewart Island robin *Petroica australis rakiura*). All races have a grey back and upper breast, and a white underbelly. There is subtle sexual dimorphism, older males (15+ months) being dark grey to black on the back, and young birds and females being a lighter grey-brown.

While robins are still found on the main islands (or "mainland"), they declined from most of their former range following European settlement. In the North and South Islands, much of this is simply due to the original forest cover being cleared. However, robins are also absent or patchily distributed in forested areas over much of their range. The species is classified as "regionally threatened", and can be considered to be in the middle tier in terms of conservation urgency. Among extant New Zealand forest birds, the upper tier is occupied by birds that are extinct on the main islands, including kakapo *Strigops habroptilus*, hihi *Notiomystis cincta*, saddleback *Philesturnus carunculatus* and little spotted kiwi *Apteryx oweni*. The lower tier is occupied by species that are still found in most forested areas within their former range: fantails *Rhipidura fuliginosa*, grey warblers *Gerygone igata*, tui *Prosthemadera novaeseelandiae* and ruru *Ninox novaeseelandiae*. The middle tier is occupied by species that, like robins, are still found on the mainland, but have declined over much of their range and are probably still declining, including brown kiwi *Apteryx australis*, weka *Gallirallus australis*, kaka *Nestor meridionalis*, kereru *Hemiphaga novaeseelandiae*, kokako *Callaeas cinerea*, tomtit *Petroica macrocephala*, rifleman *Acanthisitta chloris*, whitehead *Mohoua albicilla*, yellowhead *Mohoua ochrocephala* and bellbird *Anthornis melanura*.

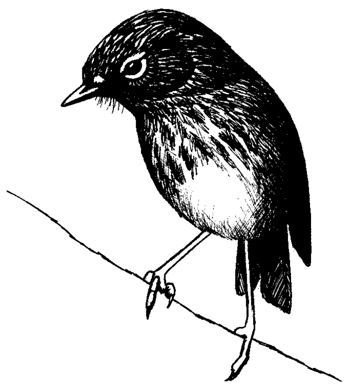
While none of the middle-tier species are rare enough to be considered critically endangered as yet, they are receiving progressively more attention. One reason for this is the idea that recovery programs are best put in place when species are still moderately abundant, which can be extended to a triage concept where most resources are invested in recovery of middle-tier species. Another reason is increased emphasis on research, with longer-term goals, use of species where reasonable sample sizes and/or experimental manipulations are possible, and testing of general principles that can be extrapolated beyond the focal species and system. Perhaps the most important reason is increased emphasis on ecological restoration, creating incentive to re-introduce any species that has disappeared from a target area even if that species is not endangered. There has been a strong emphasis on restoration of offshore islands for 15-20 years, but there is also now a growing emphasis on restoring selected areas on the mainland, mainly through intensive predator control.

Re-introductions of robins are a key component of restoration programs for several reasons. Because they are not considered endangered, robins are a low-risk species to re-introduce and have therefore been the first bird released in some restored habitats. Associated with this, robins have been able to "hang on" in some mainland areas where many other forest birds have disappeared, and can therefore be used to "test" an area before releasing endangered species. Robins are the easiest bird in New Zealand to find, observe, and capture. They are therefore an excellent species for research, and can be used to assess effects of management such as predator control. They are diurnal and friendly, hence are a bird that members of the public can see and enjoy. Finally, they are a key species in their own right, being New Zealand's only specialist ground-feeding forest passerine.

History of New Zealand robin re-introductions

The first translocations of New Zealand robins were in 1972-73, and involved South Island birds (Table 1). The rationale for these was to trial capture and transport methods to be used with black robins, and assess whether a robin population could be established from a small number of birds. Two of these translocations, to Motuara and Allports Islands, successfully established populations. Both islands had advanced regeneration following earlier human impact, and neither had predators likely to threaten robins.

The Motuara and Allports populations were both established from five birds, giving researchers an opportunity to assess bottleneck effects. Ardern *et al.* (1997) found that the Motuara and Allports populations had lower levels of genetic variation than their source populations, and Byrne (1999) found that they had lower hatch rates than their source populations. Maloney and McLean (1995) and McLean *et al.* (1999) used these populations to study responses to unfamiliar predators. They found that robins on Motuara, where stoats are absent, responded: i) less strongly to a model stoat than did mainland robins, and ii) could be trained to show an improved response to stoats *Mustela erminea*, suggesting that it could be useful to train island birds before translocation to the mainland.



New Zealand robin *Petroica australis*
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Subsequent robin re-introductions have been done in the context of restoration. Robins were re-introduced to Maud Island for a second time in 1983, based on the idea that the habitat had improved since the first failed attempt, but they again failed to survive (this was unrelated to two stoat invasions that occurred on Maud, as robins and stoats were never present at the same time). Robins were re-introduced to two other islands from 1983-86, Moturoa and Moturua, both of which had regenerated somewhat following modification but had at least one exotic mammal likely to prey on robins (stoats, Norway rats *Rattus norvegicus*, and/or ship rats *Rattus rattus*). Robins disappeared from Moturoa, but have survived on Moturua in the presence of stoats and Norway rats. Robins were re-introduced to Hawea, Entry, and Mokoia Islands from 1987-91 following rat eradication. I do not know the fate of the first two translocations, but robins have thrived on Mokoia, where the vegetation has regenerated for about 50 years following clearing and there are no mammalian predators. This population was recently harvested for a second re-introduction to Moturoa following eradication of predators there.

The re-introductions to Tiritiri Matangi in 1992 and Mana Island in 1996 were the first done in the context of intensive ecological restoration, involving re-vegetation, weed control, and re-introduction of a wide range of species. At both sites, robins were re-introduced at an early stage of regeneration when there was little forest cover. We have studied the Tiritiri Matangi population intensively, focusing on the limitation imposed by the small amount and fragmentation of the available forest. Our initial research aimed to experimentally test the effect of familiarity within founder groups, based on the idea that groups released in individual forest patches would largely stay in those patches (Armstrong, 1995). There was no evidence that familiarity was important, and robins also dispersed extensively among patches. Modeling of juvenile dispersal has shown there is no isolation effect, hence the island can be treated as a single population rather than metapopulation. Nevertheless, the robins have only bred in the original fragments (totaling 13 ha) and have not used the planted areas in between. Modeling of survival and fecundity data indicates a highly stable population, with a carrying capacity currently about 65 and juvenile survival limited by the available habitat. The Population Viability Analysis model developed for

Tiritiri Matangi suggests that robins could persist for decades in as little as 2.5 ha of good-quality habitat, depending on the effects of loss of genetic variation. The model also indicates that small populations in good quality habitat can be sustainably harvested, and therefore that island populations, or intensively managed populations, may be the best sources for further translocations. This idea has recently been tested by removing birds from the Tiritiri Matangi population for translocation to Wenderholm Reserve, and measuring the effect of the reduction on population parameters. The Mana Island population has not been so well studied but has clearly thrived. Recent observations (B. Wiles, MSc thesis, in prep.) have shown that robins occur at high density in the planted vegetation on Mana, in constrast to Tiritiri Matangi.

The first robin re-introduction that was part of a mainland restoration program was to Hinewai Reserve in 1994. The reserve is regenerating naturally, but has not had predator control or any other intensive management. The idea was to assess the feasibility of using island robins (from Motuara) trained to recognize predators for reintroduction to the mainland. The birds disappeared within 6 months, although it is not clear whether or not this was due to predation.

Subsequent mainland re-introductions have all been to areas with predator control. Three of these populations (Boundary Stream, Wenderholm, Paengaroa) are being closely monitored. Boundary Stream and Wenderholm have extremely low predator numbers, facilitated by the size of the reserve (700 ha) and intensity of control at Boundary Stream, and by geography (a peninsula bounded by a highway) at Wenderholm. Preliminary data indicate that these populations have survival and fecundity similar to that at Tiritiri Matangi, and are expected to expand rapidly to fill the available habitat. In contrast, Paengaroa is relatively small (100 ha), surrounded by farmland, and has lower-intensity predator control. Most nests were preyed on in the first breeding season, and population viability appears marginal at best.

Future directions

The history of island re-introductions, and research on the Tiritiri Matangi population, indicates that re-introductions to predator-free islands should have a high probability of success if at least some forest habitat is available. This also applies to the proposed robin re-introduction to Karori Reservoir, a 200 ha fenced-off area in Wellington that will soon be free of introduced mammals. The greater challenge is in evaluating the potential to re-introduce robins to mainland areas where predators cannot be completely eradicated. This is clearly an important challenge, given the recent spate of mainland re-introductions (Table 1) that is likely to continue.

An obvious issue to address is the level of predator control (if any) that is needed for a re-introduced population to be viable. This can

Table 1. Known re-introductions of New Zealand robins

LOCATION	YEAR	RATIONALE	RESEARCH
Conway River, North Canterbury, South Island	1972†	Trial for black robin translocation	
Maud Island, Marlborough Sounds (309 ha)	1972†	Trial for black robin translocation	
Moturoa Island, Marlborough Sounds (58 ha)	1973	Trial for black robin translocation	effect of bottleneck[2,3], predator recognition[4,5]
Allports Island, Marlborough Sounds (16 ha)	1973	Trial for black robin translocation	effect of bottleneck[2,3]
Maud Island, Marlborough Sounds (309 ha)	1983†	Natural regeneration	
Moturoa Island, Bay of Islands (157 ha)	1983†	Natural regeneration	
Moturoa Island, Bay of Islands (162 ha)	1986	Natural regeneration	
Hawea island, Fiordland	1987	Rats eradicated	
Entry Island, Fiordland	1988	Rats eradicated?	
Mokoia Island, Lake Rotorua (135 ha)	1991	Natural regeneration, Norway rats eradicated	
Tiritiri Matangi Island, Hauraki Gulf (220 ha)	1992	Revegetation	effect of familiarity[6], effect of poison drop[7], population viability[8] & sustainable harvesting[9]
Hinewai Reserve, Banks Peninsula, South Island	1994†	Natural regeneration	
Mana Island, Cook Strait (217 ha)	1996	Revegetation	Effect of habitat[10]
Trounson Kauri Park, Northland, North Island	1998	Predator control	
Boundary Stream Reserve, Hawkes Bay, North island (700 ha)	1998	Predator control	
Wenderholm Regional Park, Nr Auckland, North Island (60 ha)	1999	Predator control	population viability[9]
Paengaroa Reserve, near Taihape, North island (101 ha)	1999	Predator control	population viability[11]
Kakepuku Mountain, Waikato, North Island (200 ha)	1999	Predator control	
Moturoa Island, Bay of Islands (157 ha)	1999	Predator eradication/control	
Putauhinu Island, off Stewart Island (141 ha)	1999	Cats and kiore eradicated	

† extinct, ? Fate unknown. Details of re-introductions in the 1990's can be accessed at *http://www.massey.ac.nz/~Darmstrong/nz_projects.htm*
References: [1]Flack (1978); [2]Ardern *et al.* (1997); [3]Byrne (1999); [4]Maloney & Mclean (1995); [5]Mclean *et al.* (1999); [6]Armstrong (1995); [7]Armstrong & Ewen *et al.* (submitted to NZ Journal of Ecology; [8]Armstrong & Ewen (submitted to Conservation Biology); [9]W. Dimond (MSc thesis, in prep); [10]B. Wiles (MConSc thesis, in prep) and [11]E. Raeburn (MSc thesis, in prep).

be achieved by modeling populations and estimating effects of different levels of predator control on population parameters. Associated with this, we need to take into account the negative effects that poison operations might have on re-introduced robins. Using mark-recapture analysis, we estimated that an aerial poison drop 16 months after the Tiritiri Matangi re-introduction killed 11% of the population. Predator control is clearly a key issue, as predation has been shown to strongly limit the reproductive rates of extant mainland robin populations. Nevertheless, we currently have a poor understanding of the factors accounting for the patterns of local extinctions that have occurred, and therefore of the minimum management required to restore these populations. Robins are present and even abundant in some mainland areas, but absent from others with no obvious difference in habitat or predators present. New Zealand's forest landscape is highly fragmented and modified, yet we have no information on factors such as patch isolation effects, patch size and edge effects, and interactions between habitat quality and predation rates. Research on these factors is necessary if we are to develop cost-effective mainland restoration strategies.

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The history of translocations and re-introductions of kiwi in New Zealand

Rogan Colbourne & Hugh Robertson

Introduction

The four species of kiwi *Apteryx* spp. are flightless nocturnal ratites endemic to New Zealand. They are so secretive that few *Homo sapiens* New Zealanders or 'Kiwis' have ever seen their namesakes in the wild. Even so, New Zealanders have always had a sense of pride in knowing that their national icon, the kiwi, was living safely in the secret world of the forests surrounding them. However, quite early after the European settlement of New Zealand, some people realised that kiwi were not as safe as the public of the day perceived them to be, and translocations were used in attempts to preserve them.

Kiwi and their ratite ancestors have lived in New Zealand for tens of millions of years. During this long time they evolved in the absence of mammals, except for a few small bats, and became flightless.

When Maori reached New Zealand about 1000 years ago, kiwi were common and widespread, judging from subfossil and midden records. Hunting, dogs and fire undoubtedly had an impact on them; however, the arrival of Europeans, with their associated mammalian companions and their drive to convert rich lowland forest to pasture had a devastating effect on kiwi. It is estimated that there were tens of millions of kiwi in New Zealand in 1800, but by the early 1870s thousands were being killed and exported to satisfy the demand from foreign museums for specimens and to provide skins for the European fashion market. However, the introduction of mustelids in the 1880s, in a vain attempt to control rabbits, probably had, and is continuing to have, the single greatest impact on these ancient New Zealand birds. Today the kiwi population is estimated at only 70,000 birds, and populations on the New Zealand mainland are halving every decade (Robertson, 1999).

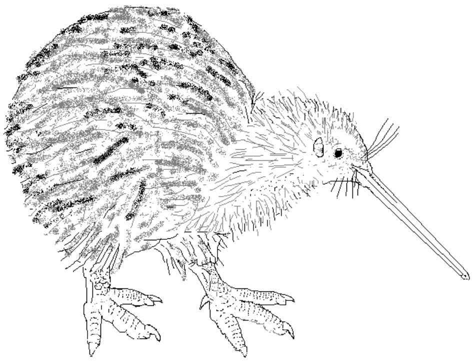
Early conservation efforts

By the late 1800s, some New Zealanders were becoming alarmed at the disappearance of many native birds, but particularly the flightless endemics such as kiwi. The first major response to this came in 1894 when Richard Henry was appointed custodian of predator-free Resolution Island in Fiordland, which had been declared a fauna and flora reserve in 1891. Henry set about translocating kakapo *Strigops habroptilus* and kiwi to Resolution Island. In three years from 1894 he transferred 474 kakapo and kiwi there, and shipped out a further 750 ground-dwelling birds to other parts of the country, but few reached their destinations through the trauma of captivity and long sea voyages.

The little spotted kiwi *Apteryx owenii*, the smallest and, originally, the commonest species of kiwi, died out on Resolution Island soon after stoats *Mustela erminea* swam to the island, and then disappeared from the mainland of New Zealand by about 1970. Little spotted kiwi has only survived because a few birds were introduced to predator-free Kapiti Island in 1912 or 1923, where they flourished and grew to a present-day population of about 1000 birds. Since 1982, they have been translocated to a further four islands (Hen, Tiritiri Matangi, Red Mercury and Long Islands), giving a total and increasing population of about 1100 individuals in 1997 (Colbourne & Robertson, 1997). In July 2000, 20 little spotted kiwi were returned from Kapiti Island to the mainland of New Zealand at Karori Sanctuary, a 250-ha water catchment area in the heart of Wellington City, which has been ringed with a predator-proof fence.

Almost all the previous successful translocations of kiwi have been to offshore islands; transfers of North Island brown kiwi *Apteryx mantelli* to several islands in the Bay of Islands, and to Kawau, Little Barrier and Ponui Islands in the Hauraki Gulf were particularly successful. One notable exception was the failure of great spotted kiwi *Apteryx haastii* to establish on Little Barrier Island after 19 birds were introduced there in 1915 (Oliver 1955).

Up until recently, there have been a few attempts to re-introduce kiwi into areas where they had previously been on the mainland. In the late 1970s and early 1980s about 70 kiwi were salvaged from areas being cleared or logged in Northland and released in southern Northland, Hawkes Bay, and the King Country. There were few follow-up surveys, little information on age classes or ratio of sexes of birds moved and, thankfully, most have all but died out. We use the word "thankfully" because recent genetic



Kiwi *Apteryx* spp.
© Rogan Colbourne

research (Baker *et al.,* 1995) has shown that kiwi populations are highly subdivided, so Northland birds are quite different (probably at the subspecies level) from those elsewhere in the North Island. In 1980-81, 41 brown kiwi were caught in Northland and released into the Waitakare Ranges north of Auckland city (MacMillan, 1990). Follow-up surveys 10 years later failed to detect any birds, although a number of unconfirmed reports were received of kiwi seen or heard many kilometres away, including in the northern suburbs of Auckland City itself. Most, if not all, of those birds caught and released were territorial adults. Recent research has shown that when territory-holding birds are caught and moved up to 10 km away they can find their way back to their original territory (J. Miles *pers. comm*). This apparent homing instinct may have contributed to the failure of the translocation, with the adult birds immediately dispersing in an attempt to return home. Of course on offshore islands, or behind predator-proof fences, this instinct is thwarted.

The kiwi recovery program

By 1990, concern was growing about the future survival of kiwi. In 1991, the Kiwi Recovery Program was launched. Its aim was to save kiwi from extinction, especially on the mainland of New Zealand. This program set out to discover the numbers, distribution and genetic variation of kiwi, the specific threats the bird faces; and to begin managing the recovery of the most endangered populations.

Data pooled from major mainland studies (McLennan *et al*., 1996) showed that adult kiwi mortality averaged about 8% per annum whereas recruitment was only 1-2%, a net rate that would see the populations halving every decade. This research showed that 95% of young kiwi die before they reach six months of age, and the blame was laid largely on stoat, and to a lesser extent, cat predation. However, it was also discovered that juvenile kiwi reach a ‘safe size’ at 1000-1200 g (about 40% of the size of an adult) at about six months old, at which time they are large and aggressive enough to fight off stoats and cats successfully. Unfortunately, kiwi of all ages are vulnerable to attacks by dogs and ferrets *Mustela furo*.

This information was used to develop two approaches for increasing the recruitment of young birds on the mainland. The first involves reducing numbers of predators immediately after kiwi have hatched and then maintaining low predator numbers for the next six months until the birds reach the ‘safe size’. The second involves temporarily removing the chicks (or removing and artificially incubating eggs) for the first six months and returning the subadults to the wild when they are large enough to protect themselves.

The second approach has been dubbed Operation Nest Egg (O. N.E.) and considerable research has been carried out under its auspices in recent years to determine if it is a practical technique for maintaining kiwi populations on the mainland of New Zealand. This work is described in the next paragraphs.

Operation nest egg

Female North Island brown kiwi usually lay two clutches of 1-2 eggs each year; however, they will often re-lay more frequently (up to seven eggs in a season) if a clutch is destroyed by a predator, or lost in some other way. The hatching rate of eggs laid by captive birds was lower than that for wild birds, and so research was needed to determine how wild birds incubate their eggs. To discover this, dummy eggs with internal temperature sensors were placed under incubating North Island brown kiwi in the wild. It was found that after the second egg was laid in a clutch, the top of the dummy egg, immediately under the sitting male, was kept at 36.5 °C, while the bottom of the egg was about 9-10 °C cooler. The dummy egg was turned on average 180° per day so that areas that had been cool were warmed and vice versa. This was an interesting result, as popular belief in kiwi captive-breeding institutions at the time was that kiwi were one of the very few birds that did not turn their eggs.

By applying these findings to eggs in incubators, the success of artificial hatching has improved markedly. In the 1997-1999 seasons at Auckland Zoo, 21 (88%) of 24 fertile kiwi eggs collected (at 15-75 days of incubation) from the wild in Northland hatched. All the chicks survived to release size (>1 kg). Raising chicks is now very straightforward and chicks are put onto an artificial diet within two weeks of hatching after enticing them first to feed on earthworms. All O.N.E. kiwi are regularly inspected by veterinarians and through these health-screenings a number of diseases such as *Coccidia* and the tick-transmitted *Babesia* have been detected for the first time in captive and wild populations.

How to get these birds back into the wild has also been studied. The first step was to remove the effect of predators and territorial adult kiwi from the stress of re-introduction, to see if the chicks could cope with the transition from captivity to the wild. A predator- and kiwi-free island (Motukawanui in the Cavalli Islands group) was chosen as the first introduction site. Ten captive-bred or captive-reared subadult kiwi were released with transmitters attached and their progress followed. As expected, most immediately lost about 200 g in weight because they had more fat reserves than wild birds of the same age, but a few actually gained weight during the first week. Not only did they cope well and choose typical kiwi daytime roost sites under dense vegetation right from the start, but some paired up and the two oldest pairs have since bred on the island.

Having demonstrated that captive-raised birds were capable of making the transition to the wild, the next step in the study was to re-introduce juveniles to areas on the mainland, where they had to cope with predators and adult kiwi. A total of 30 juveniles have now been returned to the Northland sites where the eggs or young chicks were originally collected. Eleven of these birds have been killed by ferrets (a ferret also killed at least five wild adult kiwi nearby at the same time), dogs, illegally set possum traps, or died from other natural causes. The oldest have been in the forest for almost four years and some have paired with wild birds, and two males bred in the 1999-2000 season. Their survival rate (0.66/ annum) is slightly lower but not significantly different from that (0.80/annum) of wild-bred juveniles of similar age in the same area. At Tongariro, in the central North Island, the O.N.E. program has had even greater success, with only three deaths out of the 13 captive-reared juveniles released into the wild.

The greatest cause for optimism comes from the results of releases at Okarito, in the South Island. Okarito brown kiwi (*Apteryx mantelli* 'Okarito') are problematic for an O.N.E. program as, unlike North Island brown kiwi, adults will not tolerate new juveniles in their territories and will promptly kill them. The approach here was to make the juveniles more 'bushwise'. Rather than raising the birds in pens, the young birds (4-8 weeks old) have been released onto a small stoat-free 'crèche' island where they have learnt to interact, feed and roost in a natural way. As a consequence, once they were returned to Okarito adjacent to territorial adult birds, these young birds settled back into the forest very successfully, with less initial weight loss and less imprinting behaviour than captive-reared birds. Over the past three years, the 25 juveniles returned via O.N.E. have raised the Okarito population by about 16%, although none has yet bred. Using islands as crèches is also more cost-effective than raising young in captivity because it eliminates the need for pens, artificial food, and keepers' time, so that for the same amount of money more kiwi can be raised and returned to the wild. Trials with North Island brown kiwi chicks have since shown that they can be successfully released onto an island crèche at only eight days old.

Conclusions

The five main conclusions from the O.N.E. experiments to date are:

- Greatest success in this intensive 'hands-on' program has come about by replicating the natural patterns of kiwi incubation and chick-rearing.
- Captive-reared kiwi can cope with the transition back to the wild, especially after being raised in a natural crèche environment. No released bird of over 1000 g has been preyed upon by a stoat or cat. The site chosen for release should, however, be managed through pest control and advocacy, especially where kiwi populations are near human settlements.
- Re-establishing populations in areas where kiwi used to live or where the population is severely depleted can be done rapidly with O.N.E. programs and without the problems associated with removing and translocating adults.
- A disadvantage of the O.N.E. program is that it avoids rather than solves the mammalian predation problem. It should be seen, perhaps, as a method of 'buying time' for kiwi populations until effective wide-scale control of predators (especially mustelids) is available.
- Because of its 'hands-on' nature, the O.N.E. program has captured the imagination of New Zealanders, and helped gain funding for the larger campaign against the many agents causing the demise of kiwi and other native wildlife. The Bank of New Zealand and its customers have contributed over $US 1,500,000 in the last eight years (including funds raised from special promotions surrounding the O.N.E. program). The findings of the Kiwi Recovery Program contributed to the New Zealand government announcing in 1999 the allocation of $US 3,200,000 over the next five years for research into the control of stoats, and announcing in June 2000 the allocation of an additional $US 5,000,000 for further kiwi recovery efforts.

The hope is that new and effective methods for predator control will allow kiwi populations to recover *in situ* on the mainland of New Zealand, without the need for intensive intervention like O.N. E. The effort will then turn to re-introducing kiwi to various parts of their former range on the mainland, rather than using the last resort of translocating kiwi to 'safe havens' on offshore islands.

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WEST EUROPE

Reconstruction of bird fauna in the Bavarian Forest National Park, Germany

Wolfgang Scherzinger

Introduction

The Bavarian Forest National Park was founded in 1970 in mountainous landscape along the border between Bavaria (Germany) and Bohemia (Czech Republic). The protected area was enlarged from 130 km^2 to 240 km^2 and is totally wooded. Three main tree communities are important for the fauna: pure spruce stands in the cold depressions of the valleys, near 700-800 m a.s.l.; pure spruce stands on rugged mountain ridges, averaging 1200-1400 m a.s.l.; and rich stands of mixed forest, with beech, spruce and fir; in the warmer slopes in-between. The area has a cool climate with high rainfall and a compact snow cover which persists for 5-6 months. The area was uninhabited by people and commercial forestry started only 150 years ago. This is late in comparison with forestry practices in other European countries. The base for this national conservation area are extensive woodlands, used and logged, but at a relatively high level of "naturalness", with patches of primary forest and undisturbed moorlands. Larger vertebrates such as lynx, wolf and brown bear vanished in the 19th to 20th century. Capercaillie *Tetrao urogallus* were endangered by forestry and the black grouse lost due to a marked change in agriculture practices.

From the beginning, one of the maingoals in this national park was re-introduction of important bird species through captive breeding and release. Breeding facilities were constructed for some bird species and specific techniques were developed for rearing, release and management of young birds.

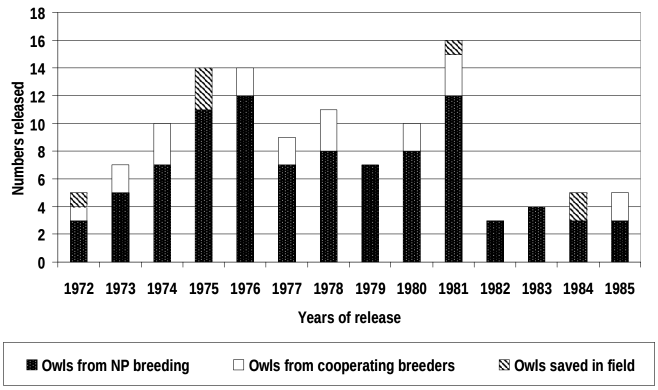
Eagle owl

The project started with the eagle owl *Bubo bubo* in 1972. Three breeding aviaries were built, two were sited in rocky areas of

Ural owl *Strix uralensis*
© W. Scherzinger/NPV

former breeding grounds and one in a zoopark which was open to visitors. In a special breeding center we produced mice and rats, which were offered freshly killed, to feed the owlets. In this captive breeding center we reared 93 young eagle owls and in addition, received 27 juvenilles from private breeders or from rescues in the field. Between 1972 and 1982, we released 100 eagle owls; 16 of them were found dead, and 12 were injured or caught as shown in figure 1. Dispersal varied from 5-124 km; owls from aviaries in undisturbed, former breeding habitats migrated much shorter distances (mean 8 km) than owls from the zoopark (mean 50 km)! Allthough some pairs settled to breed in the region, we cannot add this species to the bird-list of national park, as the owl seems to avoid the higher altitudes due to the long lasting snow cover (Scherzinger, 1987).

Fig. 1. Release of eagle owls born in captivity and those rescued from the wild: 1972-1985 (n=120)

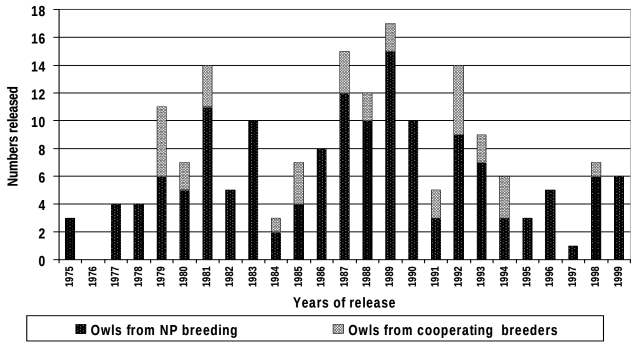


Ural owl

Another high-profile species, the Ural owl *Strix uralensis*, occured in the mountains in the 19th century but became extinct 80 years ago. This is a powerful forest-dwelling species, which preys on small mammals but also takes woodpeckers, small owls and squirrels. As there was no possibility to restore the local subspecies, we imported zoo-born owls from Scandinavia and enlarged the founder gene-pool by exchanging owls. The project started in 1972, and although little was known about their breeding we managed to construct five breeding aviaries in the national park. We gave some pairs to private keepers and zoological gardens. To support nesting in the area we set up 50 big nest-boxes of the same size the owls used in captivity.

Between 1975 and 1999, we set free 186 Ural owls (all birds were banded and some had radio-tags on tailfeathers) as shown in figure 2. Dispersal of young usually starts in October and scatters the owls over 5-40 km distances. Older owls (>1 year) did not leave the area but settled nearby. A combination of annual releases comprising both young and old birds yielded the best results. From monitoring these owls we learned that the species prefers warmer slopes with old beech-stands and it seems to be restricted to large openings in the forest. The first breeding

Fig. 2. Releases of Ural owls born in captivity: 1975-1999 (n=186)



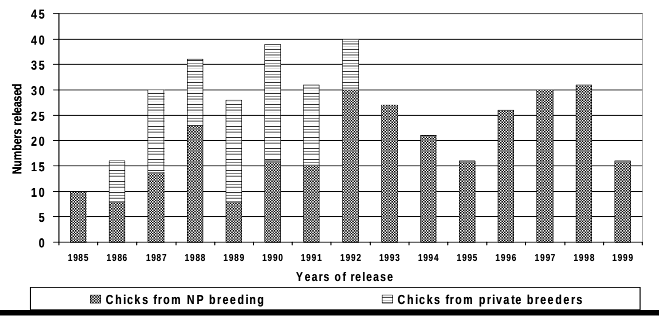
success in the wild was in 1989 (four juveniles) in the vicinity of a large windblown-area. Presently we have 5–6 breeding pairs in the national park, but are not sure of the total numbers of Ural owls present. These free-ranging pairs produce offspring very irregularly, but in years with a peak in small mammal-populations we recorded 10 owls in the field (Schaffer, 1990 & Strurzer, 1999). Nevertheless, the population is far too small and the carrying capacity of the reserve seems to be reached. There is a plan to expand the project to the neighbouring countries of the Czech Republic and Austria (Kloubec, 1997 & Steiner, 1999).

Capercaillie

The most extensive experiments were undertaken with the big cock of the woods, the capercaillie *Tetrao urogallus*, which shows a rapidly declining population in the national park (60 birds in 1974 to 12-15 birds in 1985). The goal was to release 396 captive bred capercaillie in to the national park between 1985–1999, as an augmentation exercise rather than a re-introduction. Our breeding stock of 3-4 cocks and 6-12 hens which had produced 300 chicks. In addition, 100 juveniles from private breeders were used, as this was more cost effective. Birds were banded and fitted with radio transmitters. It was difficult to create adequate conditions in which to rear birds which were fit for release. We discovered that constant training is necessary from the second week of life. The chicks must be able to recognise feeding plants, to react to predators and ultimately to be capable of brooding and rearing their own chicks. From this aspect we had to stop the cooperative program as most private birds were poorly trained and upon release either starved or were killed (Scherzinger, 1991).

Capercaillies are extremely sensitive to environmental changes, disturbance and predators, and also have a long learning period. Capercaillie are kept in the forest for two months in a 600 m² aviary which is fenced with soft netting. Releases take place at the end of October, when the birds are 3½ months old, their plumage complete and the amount of daily food is reduced. This brought an increase in survival up to 50-70% for the first winter and a large number of released birds integrated into the depleted wild population and bred sucessfully. Our project was able to halt the population decline as shown in figure 3. In the long-term we are

Fig. 3. Release of capercaillies born in captivity: 1985-1999 (n=396)



not sure if the species will survive in the national park. Large areas of natural habitat are being degraded or even lost due to forest die-back being caused by bark beetles.

From our experiences from the various release projects, we learned that combining ecological knowledge, with good techniques for breeding, rearing and handling these birds is an effective means to save these birds from extinction.

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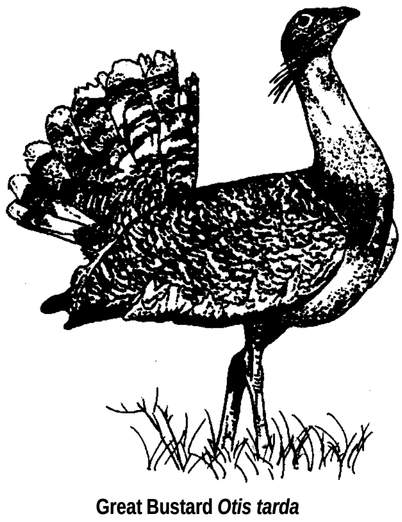
The great bustard conservation project in Germany

Heinz Litzbarski

Introduction

The German great bustard *Otis tarda* conservation project focuses on the improvement of the ecological conditions in the great bustard's habitat and on artificial breeding, rearing and the release of young birds into the wild.

The project began in 1974, at a time when 850 great bustards in 28 population groups existed in Germany (1949: 4000 individuals). In Germany great bustards live entirely in areas of intensive agriculture. Due to this, 60—90% of the eggs are disturbed and thereby lost annually. To minimise this loss a state-run program collects the eggs found by the farmers, artificially incubates them and rears the young bustards. From 1974 to 1978, in one bustard area near Zerbst (Sachsen-Anhalt), a total of 188 hand-reared young bustards were released into the wild, and 20 successful breeding attempts had been recorded by 1981. However, because the ecological conditions for the great bustards in this area only improved minimally, the population remained without young in the following years despite frequent breeding. By 1994/1995 only 3—5 birds remained, and today they are extinct in this area.



Great Bustard *Otis tarda*
© Marie-Ann D'Aloia

From 1979 to 1997, in the protected area of Havelländisches Luch (Brandenburg) a total of 288 young bustards were released into the wild. This was followed in 1998-1999 by the release of 28 birds in a second protected area, Belziger Landschaftswiesen (Brandenburg). This area is in close contact with a third group of bustards in Fiener Bruch (Sachsen-Anhalt). In these three areas, where offspring are also regularly found in the wild, the last 62-65 great bustards in Germany are currently located.

Habitat management

The measures to improve the ecological conditions in the great bustard regions began by turning a large area (1,600 ha) over to extensive agriculture in 1988, and increasing this to 5,000 ha after 1994, spread evenly between Brandenburg's two protected areas. This was set up with the following aims a) lowering direct anthropogenic chick and egg loss by exact timing/management of agricultural work, coupled with the consistent notification of brooding hens or hens with chicks, and b) renovation of the floristic diversity for a sustainable increase of arthropods in the vegetation, as an essential prerequisite for the successful development of bustard chicks.

Habitat management on ploughed land in this, up until now, intensively worked landscape, consists of limiting the use of agrochemicals, and the breaking-up of large monotonously worked land into a mosaic of rotationally and permanently fallow land. In non-arable fields, which are also regularly used by great bustards for the breeding and rearing of their chicks, ploughing and reseeding with species-poor agricultural meadowland, as well as the use of chemical fertilisers, is forbidden in order to promote the development of species-rich permanent grassland.

Anthropogenic clutch loss after 1990 was reduced from an average of 80% to less than 10%. The results of the management measures on the vegetation and entomofauna are assessed by an extensive monitoring program. This showed that within 5—12 years, the species diversity of the flora rose by 30-50%, and the arthropod biomass in the vegetation increased by 100-200%. In moist pastureland the restoration takes longer than in the formerly ploughed land. In this way the fundamental causes of the failure to produce young during the 1970s and 1980s could be largely eliminated (Block *et al.*, 1993; Litzbarski, *et al.*, 1987; Litzbarski & Litzbarski, 1996). The improvement of conditions in bustard habitat is an essential prerequisite for the successful population increase through the release of hand-reared young bustards. The artificial breeding and rearing of young bustards, as well as their release, is wrought with problems and requires an experienced team (Litzbarski & Litzbarski, 1993 & 1999).

Reproductive success

Great bustard eggs are thick shelled and strongly pigmented, so that it is not possible by holding an egg up to the light to diagnose the brooding status and development of the embryo. This limits brood success as pathological germs from dead eggs can spread explosively in the incubator, and therefore effective brood hygiene is imperative. The hatching rate of fertile eggs is 80%, the rearing success rate for chicks is 85% (1990-1998). In the wild the great bustard shows, more distinctly than many other precocial birds, a close and long-lasting bond between the hen and her chicks. Due to this, artificial rearing during the first three weeks of life demands intensive individual chick care, to ensure healthy development. Through this arises a strong bond between the chicks and the carer, which in the second half of the chicks` rearing program is gradually reduced to prepare, step-by-step, for their release into the wild. The chicks' independence is stimulated by a gradual reduction in food thereby encouraging the chicks to fend for themselves within the rearing pens.

Result of releases

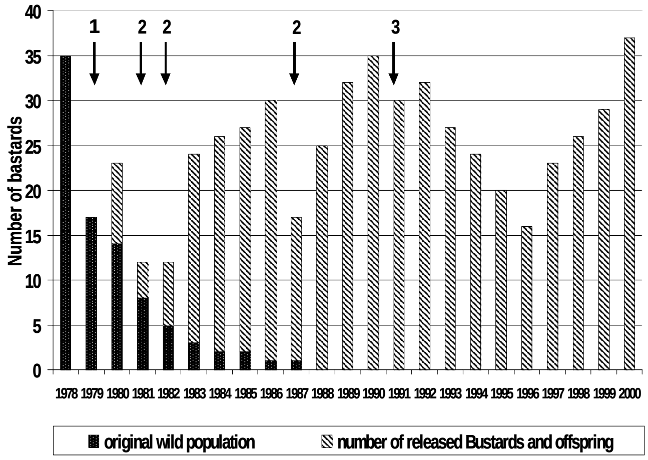
The release into the wild begins when the chicks are 8-12 weeks old. During the first 2-3 weeks they are still fed in the evenings, to bind the birds to the site and to promote the bonding amongst group members. This extra feed is stopped when the young bustards no longer search out the feeding area. In the first weeks of release it is possible, using a well disciplined hunting dog which is repeatedly sent to chase the group of young bustards to develop in the birds appropriate natural behaviours when faced with mammalian predators in the wild.

The integration of young bustards into the wild population follows in the winter of the first year and is an important prerequisite for their survival. It is essential for the success of this month-long development that sufficient grazing places exist in the release area, thus ensuring that there is no emigration from the area and allowing maintained contact between the groups. The released young birds prefer to join groups of males before the next breeding season. The losses during the first five months after release through predation (fox, goshawk) and accidents (power lines, fences etc.) account for at least 25%. Despite the losses of artificial broods, this rearing and release method produces one great bustard for every 10 eggs by the end of the first calendar year. Whereas the high losses to predation in the wild population, mean that 40 eggs must be laid for every one young bustard surviving.

During the course of the release process around 10% of the young bustards fail to lose their imprinting on people. These birds, usually males, are caught and held in a 10 ha pen and, since 1987, are used for breeding. So far no abnormal sexual behaviour has been shown by these remaining tame birds. Due to fidelity to their birthplace, hand-reared bustards settle in the release area, with only a very few exceptions. In the field the released birds display normal reproductive behaviour. The hens (often as early as their second year of life) take part fully in breeding. They have an egg fertilisation rate of 80-90%, and annual reproductive success rate of 0.1 bustards/hen. These rates are identical to those of bustard groups which do not recruit from the release program. The genetic status of the German great bustard is, despite its low population size, quite good (Pitra *et al.*, 1996).

The population development of great bustards in the Havelländisches Luch protected area indicates the importance of annual releases for their preservation, as shown in figure 1. The reproductive rate of 0.1 bustards/hen is not sufficient to maintain the population. At present the frequent release of young bustards is absolutely necessary for the preservation of the population (Streich *et al.*, 1996). Although after 1990 the high rate of chick loss due to agricultural practices was greatly reduced, at present 60-100% of first clutches are still lost through predation (corvids and other wild predators). Chicks older than 20 days also show losses of 50-100%, even without the impacts of agriculture. Amongst the corvids the non-breeding populations of ravens *Corvus corax* have clearly increased. At the beginning of the breeding season they are frequently present at great bustard nest sites and have been observed stealing bustard eggs. Hooded crows *Corvus cornix* have also been seen to attack brooding hens

Fig. 1. Population growth of great bustards in the "Havelländisches Luch" protected area



1 - beginning of conservation project, 2 – flight from winter & 3 – drastic increase in fox population

and to steal their eggs. Since 1990 populations of wild predators, have, due to reduction of hunting and immunisation against rabies, markedly increased; for example red fox *Vulpes vulpes* numbers are up 100-200%. There has also been an increase in populations of the racoon dog *Nyctereutes procyonoides,* which has colonized agricultural areas from the east in the past few years.

Research into the effects of these predators on the ecological structure of protected areas is currently underway. Reducing populations of corvids, foxes and racoon dogs in order to save species in danger of extinction is a controversial topic in Germany, and is not currently practised in any protected area.

*Contributed by **Heinz Litzbarski**, Dorfstraße, Nennhausen, Germany*

Re-introduction of griffon vultures in France

Francois Sarrazin et al.

The conservation of vultures populations is a major task in many countries through the world and particularly in Europe. In this area, the bearded vulture *Gypaetus barbatus*, black vulture *Aegypius monachus*, Egyptian vulture *Neophron percnopterus* and Griffon vulture *Gyps fulvus* are candidates for re-introduction programs. The Griffon vulture has a high conservation status and are long-lived colonial birds. They are exclusively scavengers, feeding in large groups, on mammal carrions (mainly sheep and cattle). During the last century, Griffon vultures were widely distributed in the western paleartic particularly around the Mediterranean area, southern Germany and Poland. They declined dramatically at the end of the 19th century and first part of the 20th century. Extinction causes varied according to countries but it appears that direct and indirect persecutions (shooting and poisoning) as well as reduction of carrion availability due to changes in farmland practice and veterinary laws. In France, Griffon vultures were extinct in the Southern Alps at the end of the 19th century although breeding attempts have never been formally confirmed in this area. They were extinct in the Grands Causses area in the southern Central Massif in 1945. Pyrenean populations declined similarly and only 20 to 30 pairs remained in France in the 1960s. In the Ossau valley around 10 pairs were breeding during the 1970s. This colony strongly increased after 1981 to reach more than 100 pairs in the late 90's. This population is now the biggest of the French Pyrenees, which currently contain 500 to 600 pairs. The Spanish populations decreased and increased in the same way and now the whole Spanish population appears to contain more than 17,000 pairs. Other populations remain in Croatia, ex-Yugoslavia, Greece, Turkey and Israel. The Sardinian population is declining fast and the status of Griffon's in Morocco and Algeria is either extinct or highly endangered.

In France the management of this species is to restore a meta-population located between other populations of the species in Southern Europe in Spain, Italy and Croatia which in turn are likely to be connected to more eastern populations. In addition to the natural group of colonies located in middle and western part of the French Pyrenees, five re-introduction programs have been set up to restore this metapopulation.

The first re-introduction program occurred in the Grands Causses in South Massif Central. In 1968, the idea to restore a population in the Jonte and Tarn gorges in the Grands Causses area came up due to a group of French conservationists. A first release of a few immature individuals failed at the beginning of the 1970s. It was then decided to change the release strategy and to build a captive stock in the aviaries located in front of the Jonte Gorges. During that period, the 'Fonds d'Intervention pour les Rapaces' (FIR, an NGO which is now part of the 'Ligue pour la Protection des Oiseaux'-LPO-BirdLife France) and the 'Parc National des Cévennes' (PNC) were created and have been working on this program for a long time. During the 1970's a stock of 86 individuals were obtained from Spain, French Pyrenees, zoos and those seized from illegal trafficking. From 1980 to 1986, the FIR and the PNC released 61 individuals (59 marked) in that region. Contrary to the practice in many other raptor re-introduction programs, only adult birds were released from 1981 to 1983 (age 4 year, n=39). From 1983 onwards, 20 immature and sub-adults (0—3 year old) were released. Introduced birds started breeding in the wild in 1982 and, since then, released birds have established a breeding colony from which 325 young had been produced by 1999. The colony currently contains around 70 pairs and probably more than 300 birds. Birds feed mainly on sheep carcasses coming from local farms and provided at artificial feeding sites, although feeding on naturally occurring carcasses has become more and more frequent during the last few years. Three main feeding places have been used for years but in 1998, the veterinary legislation evolved to allow farmers to use their own feeding places. This could help to restore a higher variability of food availability through space and time.

Following the great success of this first re-introduction, a program was developed in the Cirque de Navacelles in the Vis Gorges 40 km far from Jonte Gorges. Using a similar protocol, the GRIVE constituted a stock of 64 birds from 1988 to 1994. These birds were mostly obtained from French and Spanish rescue centers and from zoos. Different problems occurred in captivity entailing the death of 14 individuals probably due to vitamin B1 deficiency. From 1993 to 1997, 50 birds at least three year old were released. From 1995 to 1998 only one pair has been breeding each year and during the whole period only one young was produced. After the release of the last captive birds in 1997, the number of individuals present in the release area decreased and presently no pair breed there and it seems they may have been attracted by the Jonte colony.


Griffon vulture *Gyps fulvus*

More recently three re-introduction programs started in the South of Alps. In the Baronnies, a captive stock was constituted in 1994 and from 1996, 48 individuals have been released. Presently around 33 individuals remain in the area. Three pairs started breeding in 1998, seven in 1999 and 13 in 2000. One young was produced in 1999. In October 1999, two other programs started releasing Griffon vultures. In the 'Parc naturel regional du Vercors' 14 birds have been released around 50 km north from the Baronnies. These birds have exhibited difficulties in settlement and now only a few of them remain in the release area. Another program, located in the Verdon gorges around 40 km south of the Baronnies, released 12 individuals. These birds exhibited high dispersal abilities, one of them reached Corsica 330 km far from the release area, but now eight individuals remain in the area and four birds released in the Vercors have joined them. These programs will continue until there are at least 50 released individuals.

One interesting characteristic of these programs is that they have followed the IUCN protocol advising an accurate monitoring of all released individuals. Birds were ringed in captivity or at nest for wild born birds. Using these data it was possible to study the dynamics of the population re-introduced in the Grands Causses area. We could estimate survival rates, using capture-mark-resighting methods, and reproduction parameters for the first 10 years following the releases-i.e. from 1981 to 1991/92. Adult survival rates were very high (0.987 ± SE of 0.006). A release effect on adult survival was detected (only 0.743±0.066 survival during the first year after release). Young born in the wild (less than three year old) had an annual survival rate of 0.858 ± 0.039 during their first three years. Electrocution, mostly of young, has been the main cause of mortality but no direct or indirect persecution has been observed. Age at first breeding was four and some three-year old birds exhibited breeding behavior. That was lower than previously described in this species. The proportion of birds older than four year nesting each year increased with time but was around 0.8 over the first 10 years. Pairs constituted of birds that had been kept in captivity for more than two years showed a reduced productivity during this first period. However, the productivity of released immature and wild born birds was similar to the highest values observed in natural populations in the Spanish and French Pyrenees. These parameters are being re-estimated to take into account the possible effect of density during the last years. Furthermore feathers have been collected from nestlings since 1993 in the Grands Causses and Ossau colonies and molecular sexing techniques have been used. Preliminary results show sex ratio at equilibrium and no strong effect of sex on survival rates.

Overall, the asymptotic growth rate of the population re-introduced in the Grands Causses was high. However, although some effects on the demography of released individuals were detected, it was difficult to determine if the vital rates estimates were typical of this species or affected by the re-introduction context. We therefore compared the global dynamics of this population with the dynamics of the natural colony of Griffon vultures settled in the Ossau valley. Overall breeding success was higher in the Pyrenean colony than in the re-introduced population but the demographic parameters estimated in the Grands Causses were not sufficient to explain the increase of the Ossau population, showing the possibility of immigration in this population from the Spanish populations. Since immigration was also noticed in the re-introduced populations in the Causses, the Vis gorges and more recently in the Baronnies, we studied the habitat selection strategy of Griffon vultures by comparing the Ossau and the Grands Causses colonies. For other species, it has been proposed that the local reproductive success of conspecifics would be the best cue to assess breeding habitat quality and select breeding habitat. Three major assumptions for the verification of this breeding habitat selection were fulfilled in both colonies. First, the habitat quality varied with cliffs and year. Second, the local reproductive success was positively and temporally autocorrelated. Third, the increase of the number of nesting pairs in a given cliff from year 't' to year 't'+1 was positively correlated with the breeding success in this cliff in year 't'. Colonization of new cliffs was also favored by first nests success. Interestingly, the re-introduced population showed no major differences in habitat selection with the natural one. The release of adults likely to breed quickly was therefore a good strategy to fix a breeding colony as soon as possible. However, these birds had a lower survival and breeding rate than their descent. We developed a demographic model that predicts the relative efficiency of releasing juveniles or adults for a given life cycle and accounting for possible reduction of survival and fertility of released adults. We applied the model to the case of the re-introduction of Griffon vultures. Overall, for our case, it appeared to be more efficient to release adults than juveniles, despite the observed reduction of demographic parameters following release. This purely demographic model did not integrate this habitat selection strategy that could enhance the discrepancy between both release strategies. This habitat selection strategy is now used in an experiment involving dummy vultures to favor the settlement of breeders in the re-introduction program of the Vis gorges.

The success of the re-introduction of Griffon vultures in the Grand Causses area was mostly due to the great acceptance of the return of this species by the local human population showing the efficiency of the education program run during the 1970s. This re-introduction has been the starting point of the restoration of a metapopulation of this species in southern France but it also favored the return of the black vultures re-introduced in the Causses in 1992, and of the Egyptian vulture which came back naturally to breed in the same cliffs.

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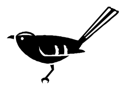

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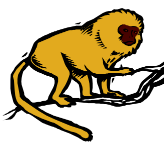

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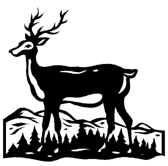

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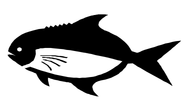

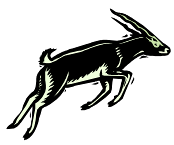

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