Endangered Species UPDATE

November 1990, Vol. 8 No. 1

THE UNIVERSITY OF MICHIGAN
School of Natural Resources

Special Issue

Captive Propagation and Reintroduction: A Strategy for Preserving Endangered Species?

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Endangered Species
UPDATE

A forum for information exchange on endangered species issues

November 1990
Vol. 8 No. 1
Suzanne Jones..........Editor
Dr. Terry Root..........Faculty Advisor
Jon Jensen................Staff Advisor

Instructions for Authors:

The Endangered Species UPDATE welcomes articles related to species protection in a wide range of areas including but not limited to: research and management activities for endangered species, theoretical approaches to species conservation, and habitat protection and preserve design. Book reviews, editorial comments, and announcements of current events and publications are also welcome.

Readers include a broad range of professionals in both scientific and policy fields. Articles should be written in an easily understandable style for a knowledgeable audience.
Manuscripts should be 10-12 double spaced typed pages. For further information please contact Suzanne Jones at the number listed below.

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Front Cover:
Red wolf (*Canis rufus*)
Photo by U.S. Fish and Wildlife Service

Back Cover:
Peregrine falcon (*Falco peregrinus*)
Photo by Mike Riley

Note From The Editor

My vision for this Special Issue began quite simply. The idea started with the observation that as the public's realization of the seriousness of today's environmental problems and the plight of a growing number of species has increased, so has the use of conservation management techniques, such as captive propagation and reintroduction, requiring resource-intensive manipulation of individuals, populations, and communities. On the one hand, I am excited about this rapidly growing body of management knowledge -- such as that concerning genetics, population viability analysis, and captive breeding protocols -- and the options that such new skills may offer to snatch back species teetering on the brink of extinction. Yet, on the other hand, I am wary of this human tendency of ours to try to solve environmental problems like habitat degradation and loss of biodiversity -- which are ultimately caused by human technology and disruption of ecological systems -- with increased human technology and even more "invasive" (from the viewpoint of the species) management methods. Thus, even while I applaud the management progress being made with endangered species, I wonder to what avail is the perfection of captive breeding techniques such as double-clutching, artificial insemination, and cross-fostering, if we still are unable to find workable resource use solutions to the problems which caused most species to become endangered in the first place.

Therefore, with both interest and reservations about the increased use of captive propagation and reintroduction, I began collecting articles from authors (many more than I originally intended!) which explore the whole spectrum of the topic: what captive breeding and reintroduction programs are currently underway, what are their goals and strategies, are they appropriate -- both in terms of effectiveness and ethics -- and finally, how well they actually are working in the field. The Special Issue begins with two articles providing general overviews of large-scale captive breeding and reintroduction initiatives for both plants and animals, followed by several pieces evaluating the effectiveness of past reintroductions, the genetics of captively breeding populations of rare species, and guidelines for determining when reintroductions are appropriate. Next comes a handful of articles addressing the appropriateness of captive breeding and reintroduction as a means of preserving endangered species, including discussions of effectiveness, measures of value and success of such programs, and ethical considerations. The issue then concludes with 18 case histories of representative bird, reptile and amphibian, plant, fish, and mammal species or species groups currently involved in captive breeding and reintroduction programs. Each synopsis briefly outlines the history of such captive breeding and reintroduction efforts and discusses the appropriateness of this management strategy in each instance.

It is my hope that this Special Issue will contribute to the international discussion concerning the appropriateness of various management strategies for preserving endangered species, and provide insights to those working around the globe to preserve biodiversity.

Suzanne R. Jones
UPDATE Editor

Special Thanks to UPDATE Supporters:

Wildlife Preservation Trust International

Wildlife Preservation Trust International is a nonprofit organization dedicated to the preservation of species threatened with extinction. This is done through a multifaceted approach of captive breeding of endangered species, research in the wild and in captivity, reintroduction of species to the wild, professional training of conservation biologists and community conservation education.

National Fish and Wildlife Foundation

The National Fish and Wildlife Foundation is an independent not-for-profit foundation established by Congress in 1984 to encourage private sector involvement in conservation by matching private contributions with federal funds. The Foundation has developed partnerships between public and private conservation organizations, corporations, individuals, and government agencies, leading to investment in over 290 projects in all 50 states, Mexico, Latin America, the USSR, and Tanzania. Major initiatives include: wetland conservation through the North American Waterfowl Management Plan, leadership training for wildlife professionals, and an annual assessment of federal natural resource agency budgets for Congress.

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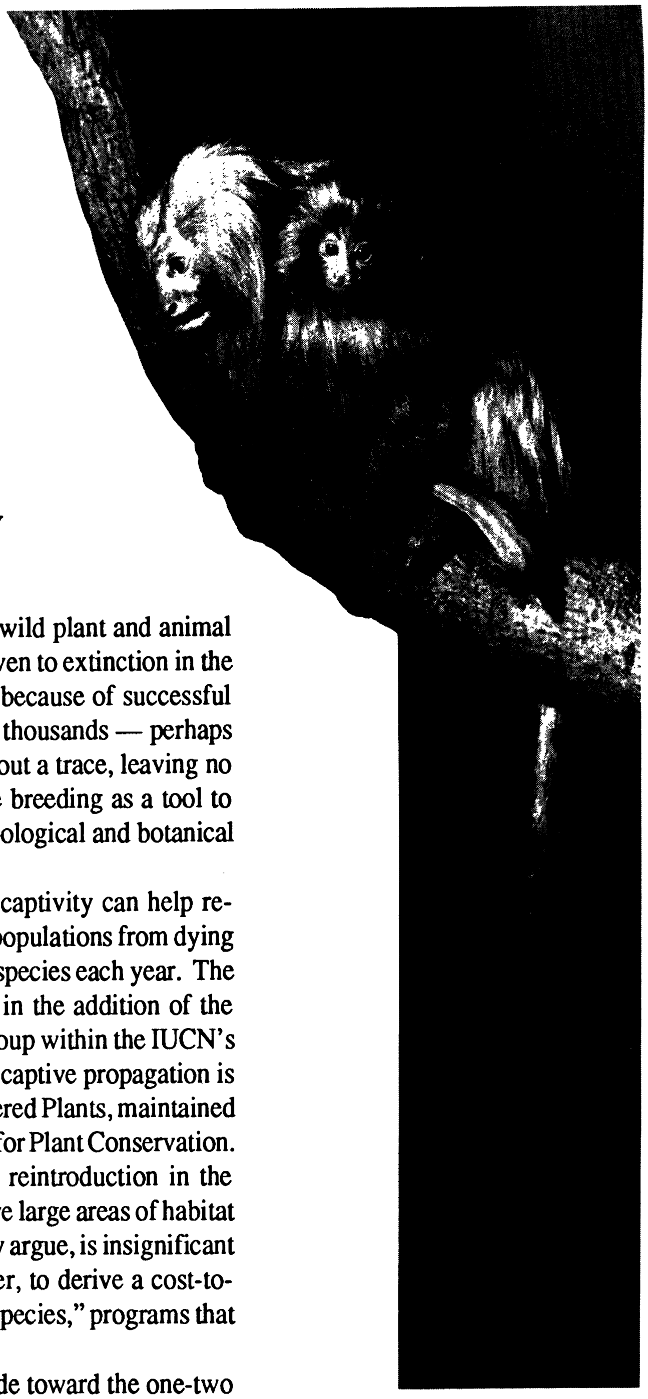
Chevron salutes the ongoing efforts of the School of Natural Resources to share the latest ideas and opinions in the urgently compelling field of conservation biology. Our continued support for publication of the *Endangered Species UPDATE* is grounded in the belief that exchanging credible information is essential for developing sound resource management strategies and making effective public policy decisions.



Captive Breeding and Reintroduction: An Overview

The idea that captive breeding can contribute to the survival of wild plant and animal species is not new. In fact, several large and charismatic animals driven to extinction in the wild within historic times survive today only in captivity and only because of successful efforts to breed them there. While they have been saved, however, thousands — perhaps hundreds of thousands — of other wild species have vanished without a trace, leaving no genetic reservoir. But can we measure the effectiveness of captive breeding as a tool to preserve biodiversity based solely upon the survival of species in zoological and botanical gardens, arboreta, and private collections?

Through reintroduction efforts, individuals born and raised in captivity can help re-establish extirpated wild populations or prevent rapidly diminishing populations from dying out. This strategy for recovery is proposed for a growing number of species each year. The increased importance of these conservation techniques is apparent in the addition of the Captive Breeding Specialist Group and Reintroduction Specialist Group within the IUCN's Species Survival Commission. Similarly, the growing emphasis on captive propagation is illustrated by the establishment of the National Collection of Endangered Plants, maintained by a consortium of botanical gardens under the auspices of the Center for Plant Conservation.

Those who would minimize the role of captive breeding and reintroduction in the preservation of biodiversity point to the larger picture, the need to save large areas of habitat and intact ecosystems. The contribution of saving single species, they argue, is insignificant by comparison and much less cost-effective. It is difficult, however, to derive a cost-to-benefit ratio for programs which focus public attention on "flagship species," programs that ultimately heighten awareness of larger environmental issues.

Although many zoo professionals have a largely favorable attitude toward the one-two punch of captive breeding and reintroduction, it has never been proposed as a panacea to prevent extinction. Neither has it been suggested as an alternative to other, more promising strategies. However, some notable successes — the golden lion tamarin, peregrine falcon, and Arabian oryx among them — demonstrate that we can use captive breeding and reintroduction programs as stopgap measures when time is short and other strategies cannot immediately be brought into play.

The task for today's conservation biologists is to determine how, when, and where captive breeding and reintroduction make the most sense in a holistic approach to conservation management. This Special Issue of the *Endangered Species UPDATE* represents an important step in that direction by putting a fair number of projects under the microscope. By considering the factors affecting the implementation of captive breeding and reintroduction efforts, as well as analyzing the progress which different agencies and organizations have achieved to date to save a variety of endangered species, we can certainly hope to develop a better yardstick for the future.

Bill Konstant
Executive Director of Wildlife Preservation Trust International

Propagation and Reintroduction of Imperiled Plants, and the Role of Botanical Gardens and Arboreta

by

Linda R. McMahan

Threats to Imperiled Plants

I find the lack of readily available information about endangered plants curious, since there are so many of them. Far more attention is given to endangered animals, even those less "glamorous" like desert pupfish and bats, than to the approximately 5,000 species, subspecies, and varieties of plant taxa in the United States that one might qualify as rare, endangered, or threatened (these are taxa ranked as G1, G2, or G3 — or the equivalent — by The Nature Conservancy). This 5,000-plus figure is particularly staggering because it represents about 20% of the native flora of the continental United States, Hawaii, and Puerto Rico.

But how many people can list ten endangered plants? Probably only those botanical specialists in private organizations and government entities who actively work with the species themselves.

This lack of information by itself is one threat to the future survival of plant species, as a lack of information also breeds a lack of vocal advocates for plant conservation. But the other, more tangible threats which occur are basically the same as for animal taxa. Such threats include human conversion of wildlands to urbanization, farming, water projects, and mining, as well as the less direct encroachment into wildlands by exotic flora and fauna and introduced diseases, and interference with natural succession. In addition, outright collecting affects many of the showier species, and is particularly harmful for slow-growing plants such as cacti. At least one rare species, *Astragalus agnicidus*, was nearly extirpated as a pest, presumed to be toxic to sheep.

Tangible threats are further complicated by two general types of rarity. (I am oversimplifying this categorization due to space restrictions, but it will serve to emphasize the complex factors that conservationists face.) One overall type is natural rarity due to limitations of climate, geography, and the plant's own natural history characteristics. The other type of rarity is the increased endangerment of taxa, which were at one time more wide-spread and common, due to habitat loss. Classic examples of the latter are plants from the central Florida scrub habitat which is being developed for housing and agriculture (see related case study), along with plants from the Los Angeles basin, southern tip of Florida, and Oregon's Willamette Valley. Species from prairie remnants of the Midwest also fit this category.

"Far more attention is given to endangered animals, even those less "glamorous" like desert pupfish and bats, than to the approximately 5,000 species, subspecies, and varieties of plant taxa in the United States that one might qualify as rare, endangered, or threatened . . ."

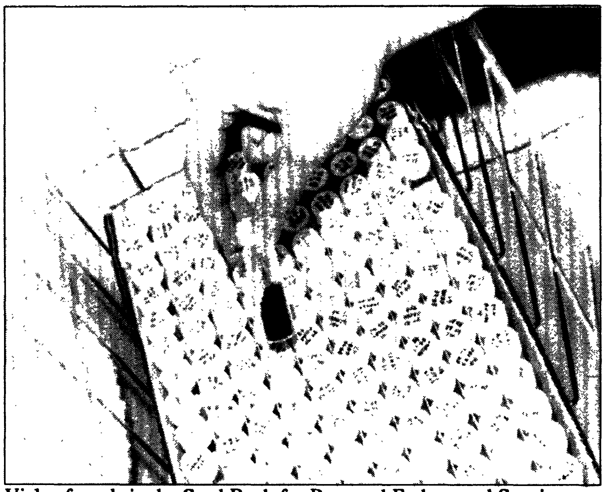
The concentration of rare species in the United States varies somewhat in location, being lowest in the Midwest and Northeast and highest in the tropical or sub-tropical areas of Hawaii, Puerto Rico, Florida, and Texas. Other "hot spots" of rarity are those areas of generally high plant diversity such as the Appalachians of the East and the coastal states of the West.

Plant conservation is closely tied to the land on which the plants grow. Plant species often have rather specific requirements, being adapted to the soil and climate of their habitat, and thus their ultimate conservation must be land-based. The extraordinary efforts of The Nature Conservancy and other similar land conservation organizations to acquire land that harbors endangered species is particularly relevant to the plants within these habitats. Although we as the conservation community do not know enough about plants and their conservation to always be effective, we do know that an essential part of what we must do is to preserve land upon which endangered plants can grow, preferably their own habitat "selected" through the evolutionary process over many generations.

Strategies Used to Conserve Plants

Despite the herculean effort to acquire land for rare and endangered plants, it has become clear over the last few decades that land acquisition alone is often not sufficient to secure a rare plant species from extinction. Other factors affecting wildlands, even protected ones, send plant conservationists scrambling to learn more about species and how to manage them. Critical needs include information about germination, growth requirements, and the basic life history information so often lacking on rare and understudied species. Information on re-establishment of rare taxa (even common ones for that matter), seems singularly lacking. Yet we must continue trying to find the answers while we are working towards conserving these same species.

The roles of botanic gardens complement those of other organizations that specialize in inventory, land protection, and legal protection. Their contribution, however, greatly exceeds the raising of public awareness about rare plants, as important as that is. What *is* most important is the basic study of rare plants, conserving their germplasm, and learning about establishment of plants in cultivation and in the wild. These tasks are not new to botanic gardens, as they are the same skills required to maintain the



Vials of seeds in the Seed Bank for Rare and Endangered Species at The Berry Botanic Garden Photo by L.R. McMahan

other plant collections in cultivation. Yet, these skills, combined with a healthy measure of scientific rigor and methods, become useful conservation tools.

The advent of the Center for Plant Conservation (CPC) in 1985 was an important step in creating scientifically credible plant conservation at U.S. botanic gardens and arboreta. The CPC, a private conservation group supported primarily by individuals and foundations, was founded as a consortium of botanic gardens (beginning with 18 and now standing at 20) to maintain offsite collections (called the National Collection of Endangered Plants) to use for conservation purposes. Gardens for the network were selected to represent as many different geographic regions of the country as possible, in order to fulfill the objective of being able to grow any species outdoors if necessary. An important criterion for selection of institutions was that each participator have a strong commitment to plant conservation. The member gardens vary from the botanical giants like the Arnold Arboretum, New York Botanical Garden, and the Missouri Botanical Garden, to small ones like The Berry Botanic Garden in Portland, Oregon, the Bok Tower Gardens in Lake Wales, Florida, and the Mercer Arboretum in Houston, Texas.

The founders of the Center probably did not realize how extensive these propagation efforts would become in a mere five years. In 1990, the 20 member gardens maintain germplasm collections of over 300 of the nation's rarest taxa, with the number growing by 50 to 100 taxa each year. Extensive cooperative efforts have evolved between the CPC and its member gardens, and federal agencies such as the Bureau of Land Management (BLM), U.S. Fish and Wildlife Service, U. S. Department of Agriculture (USDA -- including the Forest Service and National Germplasm Repository), state agencies, heritage programs, and The Nature Conservancy.

Botanic Garden Conservation Methods

Botanic garden methods vary depending upon the region of the country and the types of plants they conserve. This summary will only skim the surface of these methods, but should provide some idea of their scope and applicability.

1. Seed banks: Seed banks are good methods of conserving germplasm where they are applicable. They work best with small seeds of low oil content from temperate regions, where plants are adapted to periods of winter dormancy. For reasons I will not explain here, seed banks are less applicable for seeds which are large, oil-rich, or from tropical species. When prepared properly, appropriate seeds stored at sub-zero temperatures remain viable many times longer than they would in the wild. The loss of germplasm is slow, and if the initial sampling is large enough there is little need for breeding -- which would inevitably result in the loss of genetic diversity -- to replenish supplies. Seed banks are perhaps the only efficient method of offsite storage for annual and short-lived perennial species, because maintaining static germplasm of species which would have to be bred constantly is virtually impossible.

Many of the CPC member gardens, including The Berry Botanic Garden, maintain sub-zero seed banks. In addition, seeds are stored at the USDA National Seed Storage Laboratory in Ft. Collins, Colorado, under a Memorandum of Understanding between the CPC and the USDA. This allows storage of seed from rare and endangered plants in a variety of optimal conditions, including storage at the temperature of liquid nitrogen.

2. Living collections: Living collections are appropriate for some taxa, such as long-lived herbaceous perennials, shrubs, and trees. If space is no object, sufficient germplasm of living plants can be maintained long-term, up to several hundred years in many cases, with careful maintenance of germplasm through backup cuttings or other propagation methods. The living collections approach can also be augmented by seed bank strategies where this is appropriate. At The Berry Botanic Garden, for example, we are currently contracting with the state of Idaho to conserve germplasm of disjunct populations of western dogwood (*Cornus nuttallii*). Although common elsewhere, this dogwood is in serious decline in Idaho, possibly due to fungal infection. While botanists in Idaho work out possible conservation measures there, the Garden will store wild-collected seed and raise plants for distribution to public facilities in

Seedlings of the endangered *Sidalcea nelsoniana* in the greenhouse of The Berry Botanic Garden, being grown as part of the conservation program for the Center for Plant Conservation Photo by L.R. McMahan the Pacific Northwest to be maintained as germplasm repositories and "backup" to the wild populations.

Another plant maintained as a living collection at The Berry Botanic Garden is Barrett's penstemon (*Penstemon barrettiae*). We maintain cuttings of plants that were destroyed in lock construction at nearby Bonneville Dam. The exact germplasm is being used to re-establish plants near the site (see related case study).

3. Research: Research goes hand in hand with maintaining germplasm collections. Routine studies of germination and growth are undertaken by the CPC gardens to better understand the species in their care. Seed viability is tracked.

Increasingly, however, the CPC gardens are entering other areas of research as well. Botanists at the Rancho Santa Ana Botanic Garden, for example, have used chloroplast DNA analysis and isozyme studies to look at the taxonomic status of several rare species. This year, The Berry Botanic Garden undertook an isozyme study of the rare Gentner's mission bells (*Fritillaria gentneri*) under contract with the Rare Plant Conservation Program of Oregon's Department of Agriculture.

Gardens are also looking into the role of tissue culture in conservation. At the North Carolina Botanical Garden, tissue culture of rare pitcher plants helps establish living collections and raise plants for the specialty trade. The Arnold Arboretum is using tissue culture to try to establish disease-free plants of Florida torreya (*Torreya taxifolia*), a species in serious decline in Florida's panhandle. The Berry Botanic Garden uses tissue culture to establish disease-free living collections of endangered lilies. At some future time, it may even be possible to maintain tissue culture lines of trees and tropical species otherwise difficult to maintain as living collections. Such cultures would require much less space, but much research remains to be done before this kind of germplasm collection becomes a reality.

Other areas of botanic garden research include population demographic studies, classic taxonomic research, seed germination studies (such as the role of fire in germination), and life history studies.

4. Development of conservation policy: An area more difficult to characterize in a single phrase is the role of botanic gardens in refining conservation methods — here referred to as development of conservation policy, for lack of a better label. One example is the 1989 symposium hosted by the CPC at the Missouri Botanical Garden to learn state-of-the-art means of obtaining and conserving genetically representative germplasm from the experts in the field. A book based on the symposium is in development, but the CPC did not wait to apply information gleaned from the participants.

Student interns cleaning and processing seeds for the Seed Bank for Rare and Endangered Species of the Pacific Northwest
Photo by L.R. McMahan

From this symposium, the CPC revised recommendations regarding the amount of germplasm to collect from each population of a rare taxon, and the number of populations that should be sampled to "capture" a large part of the genetic diversity represented by that taxon in the wild. In response to these recommendations, the CPC and its member gardens have already begun recollection of some of the taxa already in collections to ensure adequate genetic representation.

5. Re-establishment of wild populations: Finally, botanic gardens are increasingly becoming involved with re-establishment of wild populations of rare and endangered species. This is a natural relationship between botanic gardens and land managing agencies, as botanic gardens are set up to propagate and grow large numbers of plants from documented sources.

Typically, a land management agency or organization such as the BLM or The Nature Conservancy will approach a botanic garden for assistance in a re-establishment program, whether it be reintroduction of a species to a former site or introduction to a new site. Currently, I would have to say that all such establishments are experimental, since we do not have a base of published or practical data on which to base these efforts; we are, in fact, learning as we go along.

After the initial agreement to undertake the effort, the appropriate seed or cuttings from nearby populations (from the exact population if material is available) are sown and grown to transplant size at the botanic garden. Sowing wild seed might work, but may take as much as 100 times the seeds used to establish plants, which is wasteful of germplasm and less effective overall.

These cultivated plants are then put into nature and receive a great deal of care, at least for their first year, including watering, weeding, and perhaps protection in wire cages to guard against predation. This care is necessary because the plants did, after all, begin their lives in cultivation, and are not particularly predisposed to survive under the whims of nature. Seeds produced by these transplants, however, are typically left to fare on their own and re-establish naturally. Often, several "pulses" of horticulturally grown plants are needed to get the population off to a good start.

When the establishment is complete, monitoring is essential to follow the population and determine the need for additional material. Similarly, managing the factors that threatened the original population is critical. Plants left on their own may easily succumb to the same threats as their predecessors.

Effectiveness of Plant Reintroduction

The question of whether plant reintroduction really works is difficult to answer because of the lack of experience to date. We can be cautiously optimistic based on a few good experiments now underway with critically rare plants. The answer will probably be that it works under certain conditions, although we have yet to determine exactly what those conditions are. We can only be certain of success for those re-established populations that are reproductive and seem to produce age structures like their counterparts elsewhere. Long-term monitoring for at least several decades will be necessary.

It is insufficient, for example, for seeds or plants to be simply placed out in the wild and forgotten. Such a project is likely to fail if the original causes of extirpation of the population or species are not controlled. Additionally, such efforts will not succeed if the plant material reintroduced is not sufficient to establish a reproductively stable population.

Successful reintroductions appear to be those set up as outlined in the previous section, i.e., those in which living plants are used and cared for in the wild, threatening factors are controlled, and ongoing monitoring is conducted. Examples include an experimental introduction — now in its fourth year— of Knowlton's cactus (*Pediocactus knowltonii*) in New Mexico, introduction of Texas snowbells (*Styrax texana*) at two sites in Texas' Edward's Plateau (see case study), and re-establishment of *Penstemon barrettiae* at Bonneville Dam.

Many factors determine success, and many considerations are important in re-establishment efforts. Such elements include maintenance of genetic diversity needed to ensure establishment, the use of local vs. non-local germplasm and the effects of germplasm origin on the establishment effort, the success of various vegetation forms (e.g., annuals vs. perennials), and similar factors.

Ultimately, of course, learning how to establish wild populations of rare plants may be essential to their survival in the wild. Susan Wallace of Bok Tower Gardens argues persuasively that maintenance of germplasm offsite for many of the species from central Florida is nearly impossible due to their poor suitability for seed storage and a typically short life span. Establishing germplasm repositories in nature, i.e., establishing introduced populations, may be the only effective way of conserving them.

Another consideration, first articulated to me by Robert Jenkins of The Nature Conservancy, is that of global warming. If global warming does in fact occur, most rare plants will be unable to survive in nature without our help. They simply will not be able to evolve or make long geographic "treks" between habitats quickly enough to survive so rapid a climatic change. In this instance, learning about re-establishment of rare plants will be essential if they are to survive.

Conclusion

One last word — in my opening paragraph I talked about the sheer volume of rare plants. The enormity of the problem causes some people to wonder if we can save them all. However, the great efforts made in the 1980s should give us all reason to be hopeful The CPC-affiliated botanic gardens have amassed a tremendous amount of rare plant germplasm in only five years. Likewise, The Nature Conservancy and others have learned of and helped to protect many habitats of rare plants, and continue to undertake this task. Federal and state agencies are seriously working toward conservation efforts, even though their progress is often frustratingly slow to us on the outside.

Although the number is high, the price-tag for conserving rare plants is low when compared to efforts needed to conserve most animal species. Seed banks and maintaining living collections offsite is inexpensive by comparison, although the cost of land acquisition is high. But even then, usually less land must be acquired to conserve habitat for a rare plant than for an animal of comparable population size. Even research, an expensive component, generally requires less money for plant than animal species. The public constituency for rare plants, even though still small, is growing. Quite frankly, I do not think it is yet time to apply the principles of triage to rare plant conservation in the United States.

Additional Sources of Information

Two good sources of information, frequently containing references and discussions on plant reintroduction and restoration include: the Center for Plant Conservation's quarterly newsletter *Plant Conservation* (CPC, 125 Arborway, Jamaica Plain, MA 02130-3520); and the Natural Areas Association's quarterly publication *Natural Areas Journal* (320 South Third Street, Rockford, IL 61104).

Specific articles are contained in:

Elias, T.E., ed. 1987. Conservation and Management of Rare and Endangered Plants. California Native Plant Society, Sacramento, CA. 630 pp.

Linda R. McMahan is Executive Director of The Berry Botanic Garden, 11505 SW Summerville Avenue, Portland, OR 97219.

Pediocactus knowltonii (Knowlton's cactus), over-collected for the commercial market, is known from only one population in northwest New Mexico. Reintroduction into an ecologically similar location has yielded survival rates of more than 75%. Photo: Peggy Olwell, Center for Plant Conservation

Overview of the Goals and Activities of the IUCN Captive Breeding Specialist Group and International Species Information System

by

Suzanne R. Jones, *UPDATE* Editor

[Regrettably, due to time conflicts, Ulysses S. Seal, CBSG Chairman, and Tom Foose, CBSG Executive Officer, were unable to meet our press deadline with their intended article on "The Role of Zoos and the Captive Breeding Specialist Group in Captive Breeding and Reintroduction of Endangered Species." (Look for an article in a later issue of the UPDATE.) Although we cannot begin to replace their insights and broad understanding of the field, this short article is an attempt to provide a brief overview of the goals and activities of the CBSG and ISIS for those who are unfamiliar with the important role which they undertake. The following material was compiled by the UPDATE Editor directly from public relations material put out by the CBSG and ISIS, and from conversations with Nate Flesness, ISIS Executive Director.]

Background Organizational Information

The International Union for the Conservation of Nature and Natural Resources (IUCN) is the largest international professional conservation organization — active in over 120 countries — in which states, government agencies, and non-governmental organizations participate equally in conservation efforts. Within the IUCN are six commissions, one of which is the Species Survival Commission (SSC). The SSC is comprised of a network of 100 specialist groups, each of which generally focus on a particular group of species, e.g., the Primate Specialist Group.

The primary responsibility of the IUCN Specialist Groups is to develop Action Plans for taxa under their jurisdiction. Action Plans are specific proposals for stabilizing and recovering imperiled species, with an emphasis on recovery *in the wild.* Specialist groups receive, at best, minimal resources from the IUCN, and are staffed and run by 3,000-plus volunteers from all over the world, serving three-year volunteer appointments. Among other things, these volunteers are responsible for procuring their own financial support to fund their conservation endeavors.

The Captive Breeding Specialist Group

One specialist group, the Captive Breeding Specialist Group (CBSG), is unique in that it cuts across taxonomic categories. The IUCN maintains that habitat protection alone is likely to be insufficient to maintain biological diversity, especially for species numbering 1,000 or less in the wild. Consequently, self-sustaining captive populations need to be established before species are reduced to critical numbers, and thereafter need to be "coordinated internationally according to sound biological principles, and a view to the maintaining or re-establishment of viable populations in the wild." The CBSG is thus charged with advising the SSC and the IUCN on the uses of captive propagation for conservation purposes, and the organizing, facilitating, and monitoring of international captive propagation programs. The CBSG primarily addresses animal species, particularly vertebrate species, within the SSC.

"... habitat protection alone is likely to be insufficient to maintain biological diversity, especially for species numbering 1,000 or less in the wild. Consequently, self-sustaining captive populations need to be established before species are reduced to critical numbers ..."

Specific CBSG objectives include the following:

1) Establish a global network of zoo professionals and zoos to provide facilities and personnel for international collaborative captive propagation programs.

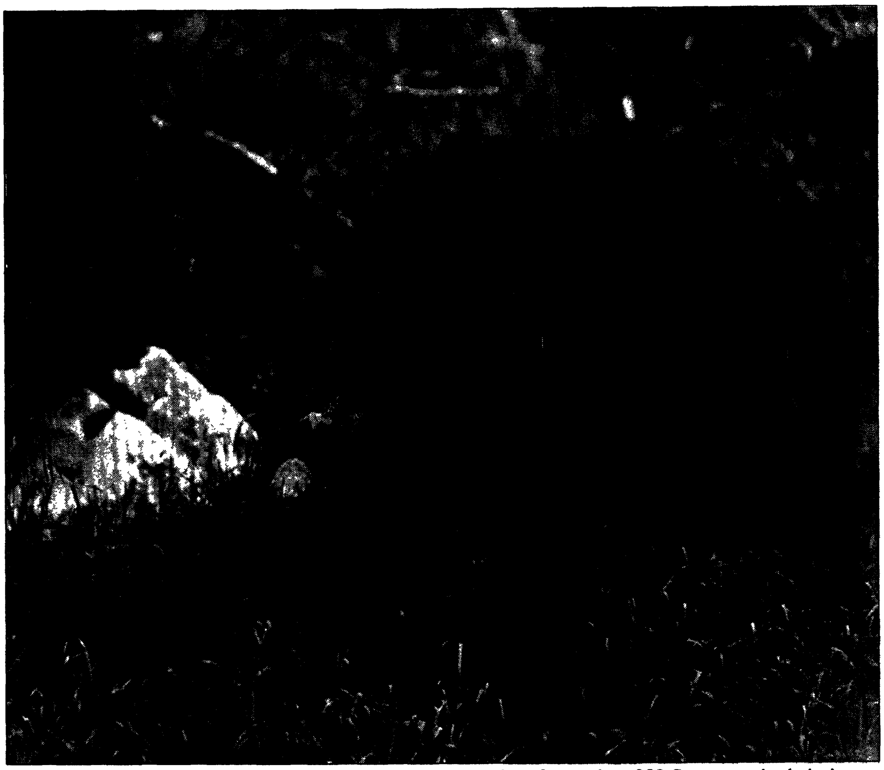
2) Establish a global network of professionals in captive and wildlife management, population biology, and other disciplines to advise on the development and conduct of recommended captive propagation programs. Collaborative programs are already underway for such species as the Sumatran rhino, kouprey, and lemurs.

3) Prepare Captive Breeding Action Plans for all of the vertebrates in collaboration with the appropriate SSC and International Council for Bird Preservation Specialist Groups. These plans are to provide analyses of the status of the species in captivity and the wild, and make recommendations of priority species for captive breeding programs. Action Plans generally focus on broader taxons than the species level. As of yet, only a handful of taxonomic orders have Captive Breeding Action Plans in some stage of development; participants are currently writing drafts for primates, parrots, carnivores, and antelopes.

Similar recovery plans are being developed by international, national and regional professional associations under the auspices of the American Association of Zoological Parks and Aquariums. These Species Survival Plans (SSP), and other equivalent international plans, require close coordination between participating institutions, e.g., for moving animals between zoos, coordinating quarantine zones, etc. Currently, there are a total of about 153 SSPs (53 in North America and an additional 100 from around the world) in some stage of development, with a goal to expand this number to 1,000. The CBSG tries to coordinate these various types of plans and efforts within the captive breeding community.

4) Assist in the organization and implementation of captive breeding programs recommended in Action Plans, including preparing studbooks, assisting in collaboration with Specialist Groups to obtain animals from the wild, conducting workshops on conservation biology and population viability analysis of individual species, and making specific management recommendations.

5) Assist in the adoption and use of effective information systems and networks, such as ISIS and ARKS, by all of the world's zoos. These information activities are essential to the work of the CBSG, and so are discussed below in further detail.



Severely endangered due to poaching and habitat destruction, fewer than 800 Sumatran (or hairy) rhinos, *Dicerorhinus sumatrensis*, remain on earth. In an emergency effort to preserve this species, four U.S. zoos (including the Bronx Zoo) and the Indonesian Government established the Sumatran Rhino Trust in 1987, to aid in rhino conservation in southeast Asia and capture doomed rhinos for captive breeding programs in Indonesia and the U.S. Photo: New York Zoological Society

Captive Breeding Information Systems

The International Species Information System (ISIS) is a global zoo animal information system which works very closely with the CBSG, and is likewise housed at the Minnesota Zoological Garden. This computer-based database contains information on over 117,000 living vertebrates, plus a greater number of their ancestors, from more than 363 zoological member institutions in 38 countries (in particular North America, Europe, Australasia, and expanding to Latin America, Asia, and Africa). Basic biological information such as age, sex, parentage, place of birth, and circumstance of death is recorded to use in various captive population analyses.

According to the CBSG, properly managed captive populations are important in providing biodiversity insurance in case species go extinct in the wild, and to prevent captive populations from being a burden on wild populations (ISIS data indicate that 92% of new zoo mammals, 71% of birds, and a "majority" of herps are now captive-bred.) The lineage information available on ISIS is particularly useful for genetic and demographic management of captive populations. Additionally, data on total captive holdings for each species allows participating institutions to better allocate available zoo space among imperiled species, and thus better fulfill their increasing conservation responsibilities.

Every six months, ISIS distributes to dues-paying members registries of "who-has-what," and short summaries of the captive status of the over 4,200 registered taxa. Such information can aid captive managers in searching for a needed specimen or locating people with expertise on a particular species. Also maintained by ISIS are partial zoological studbooks for all of the 4,200 taxa registered with ISIS. (For 95% of all captive species, no formal studbook exists, hence ISIS is the only available source for this information.)

Related systems also available from ISIS include:

1)ARKS (Animal Record Keeping System), a computer system for maintaining standardized zoological inventory records — currently used by over 250 zoos and related institutions; and

2) SPARKS (Single Population Analysis & Records Keeping System), a software tool for creating and maintaining studbooks, performing demographic and genetic analyses, and supporting population management.

One final informational activity of the captive breeding community is the preparation and distribution of the *CBSG News*. This newsletter, recently established to promote communication of news and information among people with "special interest in the uses, resources, and problems of captive breeding as a tool for the conservation of endangered species," is sent to to all members of the CBSG, chairpeople of Specialist Groups, wildlife agencies and departments with captive breeding programs, and to all of the world's zoos.

The long-term success of these numerous initiatives, and the effectiveness of these ambitious and multifaceted organizations remain to be seen. There is no doubt, however, that the CBSG and ISIS are and will continue to be pivotal contributors to captive breeding efforts to preserve endangered species.

Additional information may be obtained by writing to the CBSG or ISIS at 12101 Johnny Cake Ridge Road, Apple Valley, MN 55124, USA.

Translocations of Captive-Reared Terrestrial Vertebrates, 1973-1986

by

Brad Griffith, J. Michael Scott, James W. Carpenter, and Christine Reed

A translocation is the intentional release of animals into the wild in an attempt to establish (introduce), re-establish (reintroduce), or augment (restock) a free-ranging population (IUCN 1987), and may consist of single or multiple releases at one or several sites. An increasing appreciation of biological diversity has focused attention on translocations of rare native species to restore or preserve diversity in communities of free-ranging animals. If current patterns of habitat loss continue, natural communities may become restricted to disjunct habitat fragments. Rates of species extinction may be expected to increase in these small fragmented habitats, and periodic translocations may be required to maintain species diversity.

Several species (e.g., black-footed ferret, California condor) were relegated to captive populations as a result of systematic pressures on their populations and habitats. The ultimate goal of maintaining these captive populations is eventual re-establishment of viable free-ranging populations. Successful re-establishment of wild populations requires an understanding and effective management of all aspects of reintroductions.

Several studies analyzed factors that influenced success or failure of introductions and colonizations of non-native species (Crawley 1986, Moulton and Pimm 1986, Newsome and Noble 1986, O'Connor 1986). In an earlier paper, we (Griffith et al. 1989) analyzed factors associated with success or failure of contemporary translocations of native terrestrial vertebrates. Our analysis combined translocations of both exclusively captive-reared and exclusively wild-caught animals. In this article we focus on analysis of all translocations that included at least some captive-reared animals and compare the results to translocations of exclusively wild-caught animals.

Table 1. Translocations of native species from or within Australia, Canada, New Zealand, and the United States, 1973-1986, that included at least some captive-reared animals.

Species	Frequency	Species	Frequency
Birds (18 species)	78	Mammals (10 species)	19
Andean condor	2[a]	Arabian oryx	1[a]
Antipodes Island parakeet	1[a]	Bighorn sheep	2[b]
Bald eagle	15	Brush-tailed bettong	5
Barn owl	1	White-tailed deer	1[b]
Black stilt	4[a]	Elk	1[b]
Brown teal	6[a]	Golden lion tamarin	1
Burrowing owl	2	Numbat	1
Canada goose	13[b]	Red wolf	1[a]
Masked bobwhite	1[a]	Swift fox	4[a]
Northern black duck	1[a]	Woods bison	2
Newall's manx shearwater	1		
Nene	1[a]	Reptiles and amphibians (4 species)	6
Peregrine falcon	18[a]		
Puerto Rican parrot	1[a]		
Red-crowned parakeet	3[a]	Houston toad	1[a]
Sandhill crane	5	Johnston's crocodile	1[a]
Trumpeter swan	1	Kemp's ridley sea turtle	3[a]
Wild turkey	2[b]	Puerto Rican crested toad	1

a No translocations of exclusively wild-caught animals were conducted for these species.
b Native game animals; remainder are threatened, endangered, sensitive, or non-game surrogates for endangered species.

Translocated Species and Release Sites

In our 1987 survey (Griffith et al. 1989), we obtained detailed reports on 405 translocations of native terrestrial vertebrates from Australia, Canada, New Zealand, and the United States that were conducted during 1973-1986. Eighty (20%) of these translocations consisted exclusively of captive-reared animals and 23 (6%) consisted of both captive-reared and wild-caught animals. The remaining 302 (74%) translocations were of exclusively wild-caught animals. The purpose of translocations was similar for captive-reared and wild-caught animals: 47% to reestablish extirpated populations; 32% to augment existing populations; and 21% to establish new populations.

The 103 translocations of captive-reared animals included 32 species (Table 1). Most (56%) species were birds, and these comprised 76% of translocations that included captive-reared animals. Raptors comprised 22% of the species and 46% of the translocations among captive-reared birds. Among birds, 11% of species and 19% of translocations were native game birds; the remaining birds were threatened, endangered, or sensitive species, or their surrogates.

Mammals comprised 31% of the species and 18% of the 103 translocations that included captive-reared animals (Table 1). Ungulates, marsupials, and canids were about equally represented. Among these mammals, 30% of the species and 21% of the translocations were native game species and the remaining mammals were threatened, endangered or sensitive species (Table 1). Reptiles and amphibians accounted for 13% of the species and 6% of the translocations.

Most (73%) of the 103 translocations including captive-reared animals occurred in the United States and Canada (Table 2). At least 9 Canadian provinces or territories and 28 states released captive-reared native species during 1973-86. The remaining translocations were distributed among Australia (6%), New Zealand (14%), and other countries (8%). Captive-reared animals were released at 6 locations in Australia, 11 locations in New Zealand, and in 6 other countries.

Table 2. Release areas for translocations of native species from or within Australia, Canada, New Zealand, and the United States, 1973-1986, that included at least some captive-reared animals.

Release area	Frequency	Release area	Frequency
Australia	6	United States	49
Baird's Bay Island	1	Alabama	1
Bird Club Island	1	Alaska	3
Island A	1	Arizona	1
St. Francis Island	1	Arkansas	4
Wedge Island	1	California	1
West Australia	1	Florida	2
		Georgia	1
Canada	26	Hawaii	2
		Idaho	1
Alberta	10	Illinois	1
British Columbia	2	Iowa	1
Manitoba	2	Kentucky	1
New Brunswick	2	Maryland	1
Northwest Territories	1	Michigan	1
Ontario	2	Minnesota	1
Quebec	2	Mississippi	4
Saskatchewan	2	Missouri	1
Yukon Territory	3	Montana	2
		Nebraska	2
New Zealand	14	New Jersey	1
		New York	1
Cuvier Island	1	North Carolina	3
Huia Bay	1	Ohio	3
Matakana Island	1	South Dakota	3
Mata Pouri	1	Tennessee	3
Northland	1	Texas	2
Puke Puke Lagoon	1	Virginia	1
South Canterbury	4	Washington	1
Stanley/Stephens Island	1		
Teanau Wildlife Center	1	Other Locations	8
Tiriteri Matangi Island	1		
Wellington	1	Brazil	1
		Dominican Republic	1
		Mexico	1
		Oman	1
		Peru	2
		Puerto Rico	2

Unique Properties of Translocations of Captive-Reared Animals

Several characteristics of translocations were unique to captive-reared animals. These included source and supplementation of brood stock, source and incubation methods for bird eggs, and rearing methods.

The brood stock for translocated captive-reared vertebrates was exclusively captive-propagated for 42% of translocations, exclusively wild-caught for 27%, and composed of both wild and captive stock for 31% of translocations. In 60% of translocations of captive-reared animals, the captive brood stock had been supplemented with animals from other populations of either wild- or captive-propagated vertebrates. Most (86%) eggs yielding captive-reared birds for translocation were from captive populations, an additional 11% were from the wild, and 3% were from mixed sources. Incubation of source eggs in captivity was by mechanical means (27%), by source species (34%), by a foster species (5%), or by mixed methods (34%). In captivity, captive-propagated young were reared by their biological parents (49%), by same-species foster parents (4%), by different-species foster parents (3%), by puppets (3%), directly by humans (12%), or by mixed methods (29%). Once released, captive-propagated young were reared by their biological parents (22%), by same-species foster parents (16%), by different-species foster parents (12%), directly by humans (19%), or by mixed methods (22%). The remainder (9%) received no rearing assistance.

Comparisons Between Translocations of Captive-Reared and Wild-Caught Animals

Many characteristics of translocations of captive-reared animals were emphasized by comparison to translocations of wild-caught animals. Differences between translocations that included at least some captive-reared animals and translocations of exclusively captive-reared animals were insignificant. However, significant differences between translocations that included any captive-reared animals and translocations of exclusively wild-caught animals were numerous (Table 3).

There were differences between the kinds of animals translocated, but not between the characteristics of areas where they were released (Table 3). Translocations of exclusively wild-caught animals were in equal proportions for game species compared to threatened, endangered, or sensitive species and for birds compared to mammals. Translocations of captive-reared species were dominated by threatened, endangered, or sensitive birds. However, there was no difference between translocations of captive-reared and wild-caught animals regarding location of the release area in relation to the historic species range or release area habitat quality.

The increased care of captive-reared animals was evident in several comparisons (Table 3). At release, captive-reared animals were in better physical condition than were wild-caught animals. Habitat improvement prior to release was more likely for captive-reared animals, and captive-reared animals

Table 3. Comparison of characteristics between native bird and mammal translocations with at least some captive-reared animals and translocations of exclusively wild-caught animals from or within Australia, Canada, New Zealand, and the United States, 1973-1986. Data obtained from Griffith et al. (1989); 10 reported translocations of reptiles or amphibians were excluded from analysis. Animals that first gave birth at age 2 or younger with average clutch size of three or more are considered early breeders with large clutches; all others are late breeders with small clutches.

	Source of translocated animals				Test Statistics	
	Captive-reared		Wild-caught			
Variable	n	Percent	n	Percent	*G*	*P*
Legal Designation						
Native game species	15	15.5	148	49.7	38.8	<0.001
Threatened, endangered, or sensitive species	82	84.5	150	50.3		
Taxa						
Birds	78	80.4	154	51.7	26.7	<0.001
Mammals	19	19.6	144	48.3		
Location of release area						
Core of historic range	71	76.3	199	69.8	1.5	0.221
Periphery or outside	22	23.7	86	30.2		
Release area habitat						
Excellent	35	39.3	88	30.6	2.6	0.278
Good	41	46.1	158	54.9		
Fair or Poor	13	14.6	42	14.6		
Habitat improvement prior						
No	48	55.8	181	66.3	3.1	0.080
Yes	38	44.2	92	33.7		
Released same day delivered to area						
Yes	37	41.6	229	80.4	46.3	<0.001
No	52	58.4	56	19.6		
Hard or soft release						
Hard	27	29.7	227	77.7	69.1	<0.001
Soft	64	70.3	65	22.3		
Average physical condition on release						
Excellent	58	66.7	116	41.6	17.7	0.001
Good	26	29.9	151	54.1		
Fair or poor	3	3.4	11	3.9		
Potential competitors						
Congeneric	32	35.2	55	19.2	28.9	<0.001
Similar	36	39.6	68	23.8		
Neither	23	25.3	163	57.0		
Adult food habits						
Carnivore	49	50.5	72	24.2	40.0	<0.001
Herbivore	36	37.1	216	72.5		
Omnivore	12	12.4	10	3.4		
Early breeder, large clutch	14	14.4	127	42.6	28.1	<0.001
Late breeder, small clutch	83	85.6	171	57.4		
Result of translocation						
Failure	24	61.5	41	25.2	17.9	<0.001
Success	15	38.5	122	74.8		

were more likely to have been held on the release area prior to liberation and to have been "soft released" (food and/or shelter provided on site) than were wild-caught animals. If held on the release area, there was a non-significant tendency to hold captive-reared animals longer (14.8 vs. 2.6 days, P = 0.18) than wild-caught animals. There was also a non-significant tendency for fewer captive-reared than wild-caught translocated animals to die due to capture and handling prior to release (1% vs. 6% respectively, P = 0.164).

Release communities and ecological characteristics of translocated species differed between translocations of captive-reared and wild-caught animals (Table 3). Congeneric or morphologically similar competitors were more likely to be present in the release area for translocations of captive-reared animals. Captive-reared species also were more likely to be late breeders with small clutches or litters, were less likely to be herbivores, and were more likely to be carnivores than wild-caught species.

Operational differences between translocations of captive-reared and wild-caught animals were not always statistically different. An equivalent proportion (14%) of released animals in both groups were radio-tagged. There was a non-significant tendency for the average translocation of a captive-reared species to consist of more releases (8.2 vs. 5.5, P = 0.237) and to release more individuals (143 vs. 87, P = 0.223). The average length of the release program for captive-reared animals (3.7 yrs) was greater than for translocations of wild-caught animals (2.9 yrs., P = 0.043). The sex ratio of released animals was slightly higher for captive-reared (1.13) than for wild-caught animals (0.85, P = 0.04).

Factors Associated With Success of Translocations of Captive-Reared Animals

Translocations that included captive-reared animals were only about one-half as likely to be successful as were translocations of exclusively wild-caught animals (Table 3). However, translocations of captive-reared animals shared five characteristics that were associated with reduced probability of success among all translocations (Griffith et al. 1989). Translocated captive-reared animals were more likely ($P \leq 0.001$): a threatened, endangered, or sensitive species; a bird instead of a mammal; a late breeder with a small clutch; a carnivore or omnivore rather than an herbivore; and to have morphologically similar or congeneric competitors present in the release area than were translocated wild-caught animals (Table 3). Countering these effects, translocations of captive-reared animals shared two characteristics that were associated with increased probability of success among all translocations (Griffith et al. 1989). Translocations of captive-reared animals lasted longer and tended to release more animals than translocations of wild-caught animals.

Limited data precluded rigorous analysis of factors associated with success or failure of translocations of captive-reared animals. Only 39 of the 103 translocations could be classified as a success or failure; the survey respondents could not ascertain success or failure of the remainder of translocations at the time of data collection.

Among the 39 translocations of captive-reared animals that could be analyzed, only legal designation and food habits showed significant differences in success rates. Only 29% of the 31 translocations of threatened, endangered, or sensitive species were successful whereas 75% of the 8 translocations of native game species were successful ($P = 0.018$). The success rate was 21% for 14 translocations of carnivores, 61% for 18 translocations of herbivores, and 14% for 7 translocations of omnivores ($P = 0.022$). There was a non-significant tendency for translocations of birds to be less successful (32% of 31 translocations) than mammals (62% of 8 translocations; $P = 0.121$). However, because data were limited, expected cell frequencies were low enough to cast doubt on the statistical validity of all these univariate tests.

Limited data on translocations of captive-reared animals precluded multivariate analyses of all potentially important variables. However, the absence of differences in success rates between translocations of exclusively captive-reared and exclusively wild-caught animals in multivariate analyses (Griffith et al. 1989), permits evaluation of alternate translocation strategies for captive-reared animals with Griffith et al.'s (1989) model. Success of translocations for threatened, endangered, or sensitive species was enhanced by excellent habitat quality in the release area, releasing animals in the core of the historic range rather than on the periphery or outside, increasing the number of released animals, and increasing the number of years in which animals were released. Scott and Carpenter (1987) and Kleiman (1989) emphasized that protected and maintained habitat, identification and control of limiting factors, and proper care and training of captive-reared animals are prerequisites for successful translocations.

Appropriateness of Translocation of Captive-Reared Animals

Translocation of captive-reared animals may be preferable to translocation of wild-caught animals when a captive population exists and removal from a single wild population significantly reduces the viability of the wild population. Translocation of captive-reared animals is also warranted when IUCN (1987) guidelines are followed, and when a comprehensive analysis indicates a reasonable chance of: re-establishing a free-ranging population that enhances diversity of *native* species at an appropriate scale; establishing a satellite population that reduces susceptibility of a species to extinction from catastrophic loss of a single wild population; speeding the recovery of a species after limiting factors have been ameliorated; or effectively augmenting genetic heterogeneity of a small population with depleted genetic diversity. In the latter case, the admonitions of Greig (1979) and IUCN (1987) to avoid genetic "pollution" of disjunct populations should be heeded if possible.

Reasonable chance of success is situational. Ideally, greater than a 50% chance of success is a minimum criterion. However, a lower probability of success may be acceptable if the only alternative is to maintain the species exclusively in captivity. Acceptance of a less than 50% chance of success should not be due to releasing an insufficient number of animals. Exclusively captive population(s) should be increased prior to a translocation to provide enough animals to maximize the chance of success given the characteristics of the available release area and the translocated species.

Conversely, for species relegated to captivity, removing animals for translocation should not appreciably reduce viability of the captive population. Ideally, multiple viable captive populations should be established prior to translocation to minimize the chance of species extinction.

Except for experimental determination of limiting factors, captive-reared animals should not be released in poor or fair habitat. Estimated success for translocations of threatened, endangered, or sensitive birds to poor or fair habitat (i.e., areas where limiting factors have not been reduced) did not exceed 10% for releases as large as 400 birds (Griffith et al. 1989). Persistence of the species is enhanced more by keeping the animals in captivity and by establishing disjunct captive populations than by releasing them in unsuitable habitat.

Because release areas outside or on the periphery of historic ranges of species are associated with a marked reduction in translocation success, movement of captive-reared threatened, endangered, or sensitive species to such areas should be limited to situations without alternatives. Regardless of the number released, releasing animals in "good" habitat quality in the core of the species range is a better alternative than releasing them in "excellent" habitat quality on the periphery or outside of the species historic range (Griffith et al. 1989).

We implore people engaged in translocations of captive-reared animals to keep detailed and accurate records. Currently, we have insufficient data for a rigorous analysis of specific factors such as genetic heterogeneity, sex and age composition, or specific rearing and handling procedures that may affect the success of translocations of captive-reared animals. Only by increasing the quantity and quality of the database can we hope to fine-tune the process and increase the effectiveness of the technique.

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The Role of Genetics in Captive Breeding and Reintroduction for Species Conservation

by
Alan R. Templeton

Biodiversity is the variety of life, including variation at the genetic, species, and community levels. Today there is a crisis in biodiversity at all of these levels. The natural habitats of many species have already been completely destroyed, and those of many others have been so reduced and fragmented that the species are in danger of extinction. Even when habitat preservation is possible, it frequently requires the reintroduction of propagules. Thus, captive breeding is needed to preserve a species that has gone extinct in nature, to provide a backup for habitat preservation efforts, or to serve as a source of propagules for reintroduction.

The goal of most captive breeding programs is to maintain the species in captivity until release back into nature is possible. However, the released population will often experience an environment that is different from its original one, and all environments are subject to change, often at an increasingly rapid pace (e.g., global warming). Long-term survival, therefore, depends upon the released population having sufficient genetic variability to provide adaptive flexibility in an uncertain future. Hence, captive populations must be managed to preserve genetic diversity.

A second goal of genetic management is to alter the genetic composition of the species as little as possible from its initial state. Unfortunately, these two goals are sometimes contradictory. Captivity may induce selective forces, and populations that respond to these novel selective regimes will be altered from their wild genetic constitution. The capacity to respond to altered selective regimes depends upon genetic diversity in the population. Hence, satisfying the goal of preserving genetic diversity makes it difficult to satisfy the goal of not altering the species' genetic composition. Consequently, the genetic manager must often balance contradictory goals in the breeding program, as well as deal with practical constraints.

"... satisfying the goal of preserving genetic diversity makes it difficult to satisfy the goal of not altering the species' genetic composition. Consequently, the genetic manager must often balance contradictory goals in the breeding program..."

PRESERVING GENETIC VARIABILITY

Founder Populations

In order to preserve a species' genetic diversity in captivity, it is obviously necessary to carry over much of that diversity from the natural population into the initial captive population. The founding size of the captive population is a critical parameter in determining the extent of this carry-over, with the amount of genetic diversity increasing with increasing initial size. Unfortunately, many captive breeding programs are established only after the natural population has already been reduced to a very few individuals (e.g., the California condor), so much of the species' genetic diversity has already been lost. Moreover, with small founding numbers, the genetic manager will also frequently have to deal with additional genetic problems such as inbreeding depression (Templeton and Read 1984). Hence, rather than establish captive populations as acts of desperation, it is better to establish them as "insurance policies" when the natural populations are still sufficiently large to contain much genetic diversity.

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"Simply using a large number of founders does not ensure that genetic diversity is being preserved. Without a knowledge of how the species' genetic diversity is divided within and between local populations, it is impossible to design a sampling program that is ensured of preserving the species' genetic diversity."

Collared lizard (*Crotaphytus collaris*) Photo by A. Templeton

When the numbers in the founding population are low, the situation is more difficult genetically, but not hopeless. Nei et al. (1975) showed that populations established from a single mated pair can carry over a sizable portion of the genetic diversity found in the ancestral population when the ancestral population is panmictic (not genetically subdivided) and randomly mating. These conditions ensure that much of the species' overall genetic diversity is present in the form of individual heterozygosity. Thus, a few individuals can carry over much genetic diversity. Unfortunately, one or both of these conditions are frequently violated.

For example, my laboratory and the Missouri Conservation Commission have been involved with sampling natural populations of the collared lizard, *Crotaphytus collaris*, for reintroduction on restored habitats in the Missouri Ozarks (Templeton et al. 1990). These lizards live on Ozark glades: rocky, treeless, outcrops that provide xeric habitats separated from one another by the predominant oak-hickory forest found in the Ozarks. Hence, the natural habitat of these lizards is highly fragmented. Genetic surveys have revealed no or very little genetic diversity within a glade but extensive differentiation among glades — even glades just a few hundred yards apart (Templeton et al. 1990). Hence, even a large sample of lizards taken from a single glade subpopulation would miss almost all the genetic diversity present in this species. The best way to preserve this species' genetic diversity is to sample a few animals from many different glades, rather than large numbers of animals from a few glades.

This example illustrates the importance of genetic surveys on endangered species. Simply using a large number of founders does not ensure that genetic diversity is being preserved. Without a knowledge of how the species' genetic diversity is divided within and between local populations, it is impossible to design a sampling program that is ensured of preserving the species' genetic diversity.

Preventing Loss of Genetic Diversity

Once the founder population has been established, it is also necessary to prevent the loss of the genetic diversity that has been carried over from the natural population. Once again, population size has a great effect. The larger the ultimate captive population size, the more immune it is from the loss of genetic variation through the action of genetic drift. It is therefore optimal to make the captive population as large as is practically possible. However, there are also other parameters affecting the preservation of genetic variation.

Nei et al. (1975) showed that the loss of genetic variation decreases as the rate of population growth after the founder event increases. Consequently, it is best to increase the captive population to its ultimate carrying capacity as rapidly as possible. For example, a captive population of Speke's gazelle (*Gazella spekei*) was founded in 1969 from one male and three females (Templeton and Read 1984). From an analysis of inbreeding depression in this species, Templeton (1987) inferred that the natural population was large and panmictic. Therefore, much of the species' genetic variability would be in the form of individual heterozygosity. The population was also expanded at a rapid rate after its establishment (Templeton and Read 1984). Because of these two factors, the theory of Nei et al. (1975) predicts this captive herd should have much genetic variation. Direct genetic surveys (Templeton et al. 1987) revealed that the current captive herd is polymorphic for 14% of its loci, a figure that is typical for large grazing mammals (11.4%) or large mammals in general (13.2%) (Baccus et al. 1983). Hence, despite an extreme founder event, the herd is not depauperate in genetic variability relative to other mammals.

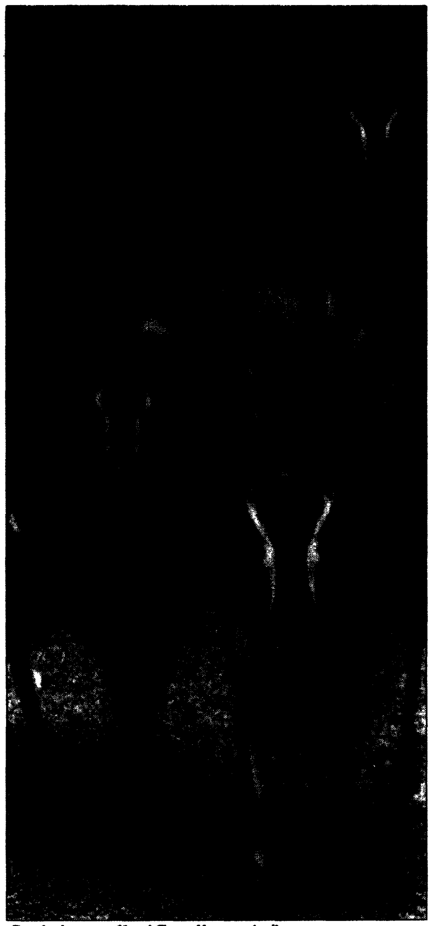
Genetic diversity is best maintained when each founder contributes more or less equally to the population. It is critical that this equalization be done from the onset of the captive breeding program. If not, much of the genetic variability present in some of the founders will be lost by chance alone. Equalizing the founder contributions in later generations can never undo this damage; once the variation of a particular founder is lost, it is gone forever. The analysis of the Speke's gazelle herd supports this prediction (Templeton et al. 1987). Before the initiation of genetic management, one founding female (F5) was bred very little during the early 1970s. Starting in 1979, the founder representation was deliberately equalized (Templeton and Read 1984), but by that time over 50% of the genetic diversity carried by F5 would have already been lost (MacCleur et al. 1986). By using the pedigree data, Templeton et al. (1987) found that none of the allelic diversity present in the herd in 1982-1985 traced back to F5 despite a fairly equal founder representation by that time. This example shows that it is critical to equalize founder representation as soon as possible after the establishment of captivity. Mistakes made early in the breeding program cannot be corrected unless additional sampling from the natural population is possible.

Once the herd has grown to its carrying capacity, genetic drift can still erode the genetic variation that has survived to that point. The best way of minimizing the impact of genetic drift in a closed population is to subdivide the population into breeding units that have minimal gene flow (Maruyama 1970). We have therefore established several subherds of Speke's gazelle at various zoos. Animals are exchanged between subherds only when they are absolutely needed for management purposes. In this way, we minimize gene flow between herds and maximize the global herd's ability to retain variation.

PREVENTING EVOLUTIONARY CHANGES

Adaptions to the captive environment may imperil the chances for success after release. As long as the captive breeding population is genetically variable, it can adapt to the captive environment. Guarding against any inadvertent selection for domestication can help, but we simply cannot anticipate or monitor all the ways in which a population can adapt to a captive environment. Hence, the goals of maintaining genetic diversity and preventing adaptation to the captive environment are sometimes in tension and complicate each other in practice.

Nevertheless, some of the strategies for preserving genetic variation also reduce the population's response to selection. For example, large founding sizes promote genetic diversity and reduce the chances of genetic transilience — rapid adaptive shifts induced by the founder event (Templeton 1980). Equalizing reproductive success in captivity to maximize the maintenance of genetic diversity also reduces the opportunity for selection in the captive population. Finally, population subdivision, the best strategy for long-term maintenance of genetic variation, tends to result in subpopulations with little genetic variation within them. Hence, there will be little response to selection under this breeding design.


Speke's gazelle (*Gazella spekei*)
Photo: St. Louis Zoo

Unfortunately, population subdivision frequently changes the genetic environment of the species. When a captive population adapts to its new environment, it is not just the external environment that is important. Because genes interact with one another, the selective fate of a gene depends critically upon its genetic environment (Templeton 1979). The genetic environment in turn is determined by the system of mating, population size, and population subdivision. When a population is brought into captivity, all three of these factors are commonly altered, resulting in adaptation to a new genetic environment. The best way to avoid adaptation to an altered genetic environment is to minimize changes in the genetic environment under captivity. To do this we need to know what the genetic environment is. This knowledge can be acquired from genetic surveys on the natural populations. If genetic surveys on natural populations are not feasible, inbreeding and outbreeding depressions can be distinguished through pedigree analysis (Templeton and Read 1984, Templeton et al. 1986), and this information can be used to infer the natural genetic environment (Templeton 1987).

"... contrary to the usual advice of no artificial selection, it is sometimes necessary to deliberately manipulate the genetic fate of the population to ensure its survival under captivity."

If the natural population is subdivided into inbred demes, one can maintain genetic diversity by sampling individuals from several local populations. One can then minimize alterations in the genetic environment by establishing subdivided breeding units, with the founders of each breeding unit ideally coming from the same local natural population. If the natural population is not subdivided or inbred, the management situation is more difficult. The best way to avoid changing the genetic environment is to avoid inbreeding within the captive population. This requires using the entire captive population as a single breeding unit. However, this strategy also maximizes the long-term loss of genetic diversity in the population and maximizes the potential of the captive population to adapt to the external captive environment. Consequently, some compromise is necessary. The fundamental decision determining which strategy to implement depends on the expected length of time the population will remain in captivity. Basic population genetic theory predicts that neutral genetic diversity is lost at a rate of 1/(2N) per generation (Crow and Kimura 1970), where N is the variance effective size. Using this neutral rate loss as an index, the breeding program should last no more than 0.6(N) generations in order to preserve at least 75% of the initial genetic diversity carried over into captivity. Moreover, by keeping the breeding program of short duration, the adaptation of the population to its captive conditions is also minimized. These considerations imply that avoidance of inbreeding is the better strategy when the breeding size is large and when the expected duration in captivity is short. The subdivided strategy is better when one can only maintain a small number of breeding individuals or when the anticipated breeding program exceeds 0.6(N) generations.

Unfortunately, practical constraints often dictate the breeding strategy much more than theoretical considerations. For example, in our Speke's gazelle breeding program, our founding population consisted of one male and three females. Once the original founders were no longer reproducing, all possible matings in this herd had to be between close relatives. Hence, there was no possibility of avoiding strong inbreeding in this captive population, so we had no choice but to opt for the inbred, subdivided population strategy.

If practical constraints or long-term considerations lead one to implement an inbred, subdivided population captive breeding strategy on an organism that is largely panmictic and outcrossing in nature, then the compromise one has to make is to adapt the captive population to high levels of inbreeding. During the course of this adaptation the population will suffer from an inbreeding depression. Such inbreeding depressions can be so severe that they may endanger the success of the entire program. Hence, contrary to the usual advice of no artificial selection, it is sometimes necessary to deliberately manipulate the genetic fate of the population to ensure its survival under captivity. This was the situation we encountered with Speke's gazelle, whose captive population developed an extremely severe inbreeding depression (Templeton and Read 1984). Since inbreeding could not be avoided, we decided to evolutionarily reduce the inbreeding depression by a breeding program, and were successful in achieving this goal (Templeton and Read 1984). This success illustrates that inbreeding depression is not an inherent attribute of inbreeding, but rather it is a genetically determined response to an altered genetic environment. Hence, as long as the captive population retains genetic variation, it should be capable of evolving a reduction in the inbreeding depression.

As this example shows, inbreeding depression can be managed. Consequently, when necessary, it is possible to implement the inbred, subdivided breeding strategy on an outbred species. By efficiently preserving genetic diversity, it is hoped that any future released population will have the evolutionary flexibility to readapt to its old genetic environment as well as to its restored habitat. To re-establish genetic variation in any future reintroduction, it would be necessary to mix animals from several subherds. It is possible that the different subherds might have evolved incompatible coadapted complexes during captivity, which in turn could lead to an outbreeding depression when the animals are mixed. To minimize the outbreeding depression, it is best to perform the initial mixing in captivity and release the healthy F_2 or later generation individuals since the brunt of the outbreeding depression is borne by the F_2 (Templeton et al. 1986). In this manner, the release population does not have to suffer from the selective pressures induced by coadaptation during captivity.

CONCLUSION

The genetic management of captive populations frequently requires compromises between preserving genetic diversity, altering the original genetic composition as little as possible, and practical management constraints. Whenever such compromises need to be made, the first priority should be given to the preservation of genetic diversity. Only if the captive population has genetic diversity does it have the adaptive flexibility that it needs to cope with an uncertain future. This same adaptive flexibility allows the captive population to evolve away from inbreeding or outbreeding depressions both in captivity and after release. Without genetic variability, the options for both the species and the genetic manager are closed. Therefore, genetic management must keep as its primary goal the maintenance of genetic diversity; all other management policies must be evaluated in terms of this central policy.

ACKNOWLEDGEMENTS

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Decision-Making About A Reintroduction: Do Appropriate Conditions Exist?

by
Devra G. Kleiman

The following example provides a framework for making decisions about whether the appropriate conditions exist to recommend (or argue against) reintroductions of captive-born animals or translocations of wild individuals or groups. Recent reviews provide more detailed background (Griffith et al. 1989, Kleiman 1989, Stanley-Price 1989); components of a good reintroduction program, as discussed in Kleiman (1989), are presented in Table 1.

Table 1. The elements of a successful reintroduction program involving captive-bred mammals. [Reprinted from Kleiman (1989), with permission from *BioScience* 39(3):154. © 1989 American Institute of Biological Sciences.]

Captive population
Ongoing research in behavior, genetics, physiology, nutrition, reproduction, and pathology
Genetic and demographic management of the population
Self-sustaining viable captive population

Field Studies
Regular censuses of the size, distribution, and genetics of the wild population
Behavioral ecology studies (home range size, movements, habitat preferences, social organizations, mating system, feeding, and anti-predator adaptions)
Locating existing suitable habitat containing critical resources for reintroduction

Habitat preservation and management
Protection of habitat from deforestation, degradation, and exploitation
Restoration and management of degraded habitats
Increase or maintenance of the number of preservation areas

Conservation education for long-term support
Professional training through academic studies, workshops, internships, courses, and fellowships
Determining the most appropriate public relations and educational strategies through surveys
Public relations educational efforts using appropriate mass media (e.g., television, radio, magazines, and newspapers)
Local community education, both formal and informal

Preparation and reintroductions of animals
Choice of candidates and assessment of their characters for retrospective correlation with postrelease survival
Training in survival techniques, including foraging and feeding, antipredator tactics, locomotion, and orientation
Adaption to local conditions at release site (food, climate and temperature, and disease)
Release and long-term monitoring to evaluate causes of death and basis for survival

The lion tamarins (genus *Leontopithecus*) derive from the Atlantic coastal rainforests of Brazil. All three species — *L. rosalia* (golden lion tamarin), *L. chrysomelas* (golden-headed lion tamarin), and *L. chrysopygus* (golden-maned lion tamarin) — are endangered, mainly due to habitat destruction and alteration. There are captive populations of each species, but at different levels of development.

Table 2 lists ten necessary conditions which should be met in order to recommend a reintroduction/translocation program. Additionally, it evaluates the position of each form of lion tamarin with respect to each condition. Finally, a general recommendation is presented concerning whether a program of reintroduction is appropriate for each form at this time.

A reintroduction is unwarranted unless the causes for the initial reduction in species numbers have been removed. The major reasons for the decline of the lion tamarins has been deforestation of the Atlantic coastal rainforest, although there has also been a thriving commerce in these forms historically. There is still significant ongoing deforestation within the ranges of *L.chrysopygus* and *chrysomelas*, thus dictating against a reintroduction at this time. It is questionable whether the reasons for the decline of *rosalia* are now fully under control.

A reintroduction is not recommended without having protected habitat available. It is likely that sufficient protected habitat is available for *chrysopygus*, but not *chrysomelas*. Protected habitat exists for *rosalia*, although in insufficient quantities for this species' future survival.

It is preferable to reintroduce individuals or groups into unoccupied habitat to prevent social disruption and disease transmission between wild and introduced individuals. Thus, areas with small or no populations of wild tamarins need to be available. This condition exists for *rosalia*, and probably for *chrysomelas*. However, the situation for *chrysopygus* is unknown at this time.

Similarly, reintroductions should be encouraged only when there is some certainty that the release of animals from different regions (both captive- and wild-born) will not jeopardize the existing native population through the transmission of disease and/or social disruption. We do not have this confidence for any of the three forms of lion tamarins at this time, due to our limited knowledge of their biology and status.

The evaluation of the success of a reintroduction can only be accomplished by long-term monitoring, and must be based on a thorough knowledge of a species' biology, distribution, and ecological requirements. On a scale of one to five, with five being the best case scenario, I suggest that there is sufficient information available for *rosalia*, totally insufficient information available for *chrysomelas*, with *chrysopygus* somewhere in between.

Local support is essential to the success of a reintroduction. A conservation education program in conjunction with a reintroduction will attract and inform the

Golden-headed lion tamarin (*Leontopithecus chrysomelas*)
Photo: Jessie Cohen, National Zoological Park

local populace, and result in greater community support for the effort. Both *rosalia* and *chrysopygus* conservation programs have strong educational components, while the education program for *chrysomelas* is just developing.

A prerequisite to the reintroduction of animals currently in captivity (whether captive- or wild-born) is a secure well-managed captive population, with a long-term masterplan and available surplus animals. A reintroduction should not jeopardize the genetic or demographic composition of the captive population. This condition is met in *rosalia*, but not in *chrysomelas* and *chrysopygus*. An exception are some wild-born confiscated specimens of *chrysomelas* in captivity in the Una Biological Reserve in Bahia that cannot be easily absorbed into the captive population.

We still have much to learn about the methodologies of preparation, adaptation, and release of lion tamarins. With so many unanswered questions about the techniques that will ensure success, e.g., for the injection of single animals into established reproductive groups, I suggest that we still consider reintroduction an experimental approach for this taxon.

The success of a reintroduction program can only be evaluated through a comparison of the behavior and survivorship of reintroduced vs. wild-born animals. Thus, access to the resources necessary to monitor the activities of released animals is essential for a reintroduction effort, especially since we have not yet perfected our preparation and release techniques. The conservation programs for *chrysomelas* and *chrysopygus* are not yet sufficiently developed, with respect to financial support and the necessary infrastructure, to warrant a reintroduction effort. On the other hand, the *rosalia* program has a well-developed infrastructure and considerable resources to monitor the activities of released animals.

One major goal of a reintroduction program is to augment the numbers or genetic diversity of a population; *rosalia* currently needs such augmentation, while *chrysomelas* does not. The situation for *chrysopygus* is not clear at this time.

Weighing the degree to which these necessary conditions are met for each species suggests that while reintroduction efforts may be appropriate for *rosalia*, they are not for *chrysomelas* or *chrysopygus* at this time. Given the limitations of personnel, money, time and resources for conservation activities, along with the limited number of surplus individuals available in populations of imperiled species, it is imperative that the need for and likely success of a reintroduction be thoroughly evaluated before it is attempted.

Table 2. Decision-making concerning the reintroduction of lion tamarins (*Leontopithecus*): Do the necessary conditions exist? (scale: 5=best; 0=worst)*

	Leontopithecus *rosalia*	*chrysomelas*	*chrysopygus*
1. The reasons for the reduction in species numbers have been eliminated (e.g., hunting, deforestation, commerce)	?	no	no
2. Sufficient habitat is protected and secure	yes?	no	yes
3. Available habitat exists with low densities or without native animals	yes	yes?	?
4. Certainty that the release of animals will not prejudice the existing wild population	no	no	no
5. Sufficient information exists about the species' biology in the wild to evaluate the program's success	5	1.5	3
6. Conservation education exists	5	2	4
7. The population in captivity is secure, well-managed, and with surplus animals	yes	no	no
8. Knowledge of the techniques of reintroduction	3	3	3
9. Resources for post-release monitoring	yes	no	no
10. Need to augment size/genetic diversity of wild population	yes	no	yes?
IS REINTRODUCTION RECOMMENDED?	**YES**	**NO**	**NO**

*Based on material provided by C. and S. Padua, A. Rylands, C. Alves, J. and LA. Dietz, F. Simon, B. Beck, and J. Mallinson at the *Leontopithecus* Management Workshop, Belo Horizonte, Brazil, June 19-23, 1990.

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Is Captive Breeding an Appropriate Strategy for Endangered Species Conservation?

by
Tony Povilitis

Is expanded use of captive breeding for species preservation in the best interest of biological conservation? The answer to this question depends on one's view as to whether humans can change their environmentally destructive ways, on how conservation is defined, and on a host of ethical and related concerns.

By and large, captive breeding proponents tend to adhere to one of the following two views regarding the role of captive breeding in conservation. According to the "last ark" perspective, human behavior will not change and severe habitat collapse is inevitable (Foose 1986, Cade 1988). Captive maintenance is thus needed to save as many species as possible, with the hope that their release back into the environment might be possible when the human population stabilizes or declines — perhaps within the next 200 to 1,000 years (Foose 1986). As Conway (1988) pointed out, "it seems inevitable that most large land vertebrates and many plants eventually will survive only as wards of humans."

The alternative view regards captive breeding as only a "hedge against extinction," an interim measure needed to retain species (Seal 1988), while factors which threaten them (e.g., habitat destruction, hunting, pollution) are brought under control. If habitat is lost entirely there is little point to species preservation (Ehrlich and Ehrlich 1981). Thus, the role of captive breeding falls squarely within the context of habitat conservation and other safeguards for species within nature.

In this paper, I argue that the limitations of captive breeding and other related considerations severely undermine the "last ark" strategy. However, captive breeding *if combined with far more comprehensive efforts to protect and restore habitat* can be of substantial benefit in species conservation.

Biological Limitations

"Last ark" conservation might preserve some 1,000 taxa of terrestrial vertebrates out of some 1,500 taxa estimated to become so endangered by the middle of the next century that captive propagation will be essential for their survival (Foose 1986). However, little help is expected for the vast majority of invertebrates and plants which comprise most of the world's vanishing biota. These taxa, which are among the most basic components of ecosystems, nevertheless dominate the 4,000 to 6,000 species per year conservatively estimated to be lost over the next decade from tropical deforestation alone (Wilson 1987, 1989).

"...the 'last ark' [captive maintenance of species] will not be able to accommodate most endangered species. It cannot foster the ecologically meaningful evolution of species. Nor has it room for higher elements of natural diversity, such as larger natural communities and landscape ecosystems."

Even endangered species successfully maintained in captivity face a precarious future. In terms of the selective forces that shape a species, captive conditions cannot simulate wild environments. When species are removed from their habitats, natural selection ends, and selection for the captive environment begins (Lyles and May 1987). When species are isolated from natural communities, coevolution of predators and prey, pollinators and flowers, biological mimics and models, etc., is curtailed. When wild animals are denied the opportunity to choose and compete for mates, sexual selection ends. Moreover, for adaptive evolution to occur, relatively large gene pools, generally on the order of thousands of individuals in the case of vertebrates (Soule 1987), should be subjected to wild selection.

Important behavioral changes in captivity have been noted even over the short term, as in the case of altered foraging behavior in recently released Mississippi sandhill cranes (Zwank et al. 1988). Special pre-release training is being developed for arboreal golden lion tamarins (Kleiman 1989), to improve locomotor and foraging skills before release into the wild. Because of uncertainty about the ability of captive-bred animals to survive in nature, some captive programs have begun propagating animals in limited wild environments, such as the propagation of free-roaming red wolves on islands for later release in mainland reintroduction programs (Rees 1989).

In sum, the "last ark" will not be able to accommodate most endangered species. It cannot foster the ecologically meaningful evolution of species. Nor has it room for higher elements of natural diversity, such as larger natural communities and landscape ecosystems. Finally, captive breeding may preclude the eventual restoration of some species to the wild by inadvertently promoting the selection of traits suited to captivity.

Drawbacks for Wild Populations

In some instances, endangered species in the wild may be harmed by increased focus on captive breeding. Removal of wild individuals to establish or stock captive populations may jeopardize the viability of seriously endangered species. For example, the recently developed Species Survival Plan for the Florida panther calls for removing ten individuals in 1990, and eight for at least the following two years (Seal and Lacey 1989), from a wild population currently numbering only an estimated 30 to 50 animals (Jordan 1990). Risks of losing individuals of more sensitive species during capture or in captivity, or by program-stimulated illegal removals could further stress some wild populations. Groups of endangered huemul (Andean deer), for example, are known by the author to have been repeatedly lost during capture attempts in the wild and through infectious diseases in captivity, with uncertain consequences for donor populations. Use of private populations to supply animals for formal breeding programs, such as that for the thickbilled parrot (Johnson et al. 1989) and angonolia (Burke 1990), could unwittingly stimulate additional taking from the wild.

Finally, wild populations of endangered species and other wildlife could be at risk from released captive-bred animals not adequately screened for diseases, and from maladaptive genes or behaviors acquired in captivity. Disease outbreaks in wild desert tortoises, for example, may have begun with the release of infected captive animals (U.S. Fish and Wildlife Service 1989). Also of concern are disease outbreaks in wildlife originating from live vaccines used to protect released captive stock.

Logistical and Financial Constraints

Some endangered vertebrates may be lost from the "ark" because of breeding failures or disease vulnerability in captivity. Endangered parrots, for example, are susceptible to both problems (Derrickson and Snyder in press). Still other species will be excluded from captive programs for purely logistical and financial reasons.

Maintaining large numbers of endangered vertebrates in captivity will be very costly over the long term. Foose (1986), for example, estimated that some $1,500 per individual animal per year is needed for conventional breeding programs, exclusive of research needs.

Huemul or South Andean deer (*Hippocamelus bisulcus*) Photo by T. Povilitis

Maintaining 1,000 vertebrate taxa would require maximal efficiency in use of zoo facilities (Foose 1986), or virtually all zoo spaces (Seal 1988). Given these limitations, Seal (1988) has characterized ex situ management of endangered species as "a temporary haven for . . . those species that happen to gain a seat."

". . . short-term successes at captive propagation provide false reassurance that the endangered species problem is solvable through ex situ measures."

Endangered species will inevitably be in direct competition for limited zoo space. Will priority be given to species on the basis of biological needs or will other factors, such as display value and other institutional needs, prevail? Will zoos vie for the most "charismatic" endangered species? As of 1986, for example, there were at least 700 captive Prezwalski's horses, with the number growing by perhaps 100 individuals per year (Smollar 1986). Moreover, large scale captive breeding programs, such as the controversial program currently envisioned for the Florida panther (Seal and Lacy 1989) could quickly fill zoo spaces, leaving other endangered species in the lurch.

As captive species become ever more dependent on biotechnology and as habitat conditions deteriorate further, conservationists face the specter of having to protect their "investments" through still more intensive manipulation and care of both captive and released animals. The high costs involved could make the whole endeavor of captive breeding and reintroduction prohibitive.

Ultimately, a massive "last ark" to preserve species would depend on sustained socioeconomic stability over centuries. But such stability appears most unlikely with continued destruction and destabilization of the world's ecosystems.

Political and Social Consequences

Captive breeding as a strategy for preserving endangered species threatens to undermine habitat protection as the central focus of conservation. Promotion of captive breeding politically, for example, could misdirect public policies away from habitat conservation and divert funds that might otherwise be used for in situ endangered species programs. At a recent hearing on legislation concerning the captive breeding of sea turtles, Representative W.J. "Billy" Tauzin (D-LA), the bill's chief sponsor, indicated that the stocking of wild populations with captive-reared animals could mitigate losses due to economic activities, as for instance, in the case of logging impacts on the northern spotted owl. In essence, preservation by such means would replace conservation of self-sustaining populations.

Captive breeding already appears to have undermined habitat conservation. In the case of the Gila topminnow, for example, stocking of artificially maintained habitats with captive-bred fish has resulted in the reclassification of the species from endangered to threatened, despite continued habitat deterioration (Simons et al. 1989). Other species, like Attwater's greater prairie chicken (U.S. Fish and Wildlife Service 1988) and several endangered river fishes (Hunt 1989), continue to decline because of habitat problems while recovery programs emphasize captive breeding. A recovery program for the Florida panther proposes a captive population of some 350 individuals which would count toward any future "recovered" population (Seal and Lacy 1989). Yet current recovery efforts for the panther lack adequate habitat evaluations and conservation plans, while the wild population of 30 to 50 animals continues to decline. In short, captive breeding too often represents the path of least political resistance for government agencies -- a detour around controversies involving land and water use.

Florida panther (*Felis concolor coryi*) with young — Photo by K. Chris Belden

As Cade (1988) noted, "most human beings continued to be captivated more by what engineers do than by what ecologists say." Indeed, short-term successes at captive propagation provide false reassurance that the endangered species problem is solvable through ex situ measures.

Will the impressionable public see endangered species in zoos merely as curious rare objects or as living beings having intrinsic worth? Will it perceive captive breeding technology as solely a tribute to human ingenuity, or as a moral obligation to temporarily offset the impact of society's misguided destruction of habitat?

Ethics

Captive breeding and manipulation of endangered wildlife raise important ethical questions for both conservationists and for society as a whole. For example, captivity denies wild animals the opportunity to experience those habitats for which they are, from evolutionary and ecological standpoints, most suited; lost is the opportunity for them to engage in a range of behaviors in response to diverse natural stimuli (e.g., foraging on a varied diet, play activities, mate selection, flexible daily routine, etc.). Indeed, habitat has value for individual animals as well as for species.

Captive maintenance, usually for reasons of economy, may also result in direct animal abuse. Wild animals are sometimes held in overly restrictive quarters, subjected to harassment while on public display, deprived of minimal sensory stimuli, sold as surplus animals to unscrupulous dealers, or otherwise treated inhumanely (Fox 1990). Captive breeding too often has more to do with justifying the existence of zoos than with preserving endangered species (Grandy 1989).

Captive propagation of endangered species also subjects animals to undue stress and mortality when used to stock wild populations that are jeopardized because of poor habitat conditions or adverse human activities (e.g., release of sea turtles where they are likely to be drowned by shrimp nets).

From a biocentric view, captive breeding of endangered species is ethically acceptable only *if* the animals or their descendants are released into reasonably secure habitats. But what if conditions continue to deteriorate and there are no certain plans to reverse the situation? Is it ethical to support captive breeding when, in deference to purely human (viz. economic) interests, society fails to take a firm and unequivocal stand in protecting critical habitats?

". . . captive breeding too often represents the path of least political resistance for government agencies -- a detour around controversies involving land and water use."

Many people find value in free-living and unmanipulated wildlife. Yet conservation efforts that depend largely on captive maintenance risk "preserving" wildlife by taming it. The result could be eventual domestication for the more "useful" captive species, and extinction of the others. In any event, captive breeding on a large scale may well destroy our sense of the beauty, mystery, spirituality, and dignity of species, and of what remains of the natural world.

Conservation Ethos

A basic tenet of conservation is to maintain in a safe and sound state the ecosystems upon which species depend, and to preserve basic ecological and evolutionary processes. This tenet is central to the U.S. Endangered Species Act, and perhaps has been best articulated in Leopold's land ethic (Leopold 1949).

The danger is in viewing captive breeding and related technologies as an alternative strategy for conserving species. Conservation in captivity as opposed to that in nature are two very distinct things. Indeed, the term "conservation" as applied to a captive breeding strategy seems wholly unwarranted, considering the tiny fraction of species that can be protected through such efforts, the problem of adverse impacts of captivity on behavior and gene selection, and the fact that nature consists of far more than species. To regard captive maintenance of a species as conservation is to commit the conceptual error of seeing that species as somehow separate from its habitat.

"To regard captive maintenance of a species as conservation is to commit the conceptual error of seeing that species as somehow separate from its habitat."

Confronted with a severe habitat crisis, conservationists need not retreat to a "last ark" ethos. But they should recognize that the "preserve what we can" approach to conservation has failed. More explicit goals are needed. One goal might be to identify and conserve on a regional basis habitats essential to endangered and declining wildlife populations, threatened natural community types, and large landscapes that can still support most native species (Noss 1983). Another might be the application of a no-net-loss principle to all sensitive wildlife habitats (Povilitis 1990). Still another goal might be to limit our own numbers, for without limits there can be no politically feasible limit to human use of land and water. Courageous and determined conservation leadership will be needed — despite institutional pressures and the risk to jobs.

Captive breeding can play a key role in any habitat-based strategy for preserving nature. But so must advances in transportation and energy use systems, in telecommunications, recycling technology, environmental education, birth control, and family, urban, and land use planning. After all, species preservation is only one reason for preserving habitat. Others include a healthful and aesthetically wholesome environment for people, and reduced risk of environmental catastrophe, such as that potentially caused by global warming. The broadest task is to integrate conservation values and techniques with economic planning and development.

An appropriate goal for captive breeding is to help preserve endangered species as part of comprehensive planning to protect and restore their habitats. There is scarce room for compromise. The focus must be on the expeditious return of endangered species to the wild, so that adverse biological, social, political, and ethical consequences of captive breeding can be avoided.

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Measures of the Value and Success of a Reintroduction Project:
Red Wolf Reintroduction in Alligator River National Wildlife Refuge

by
Michael K. Phillips

Reintroduction is an important technique for recovering endangered and threatened species (Griffith et al. 1989). Unfortunately, the technique is complex and costly (Clark and Harvey 1988), and there are few if any accepted guidelines for defining its value and success. Although these concepts will be defined in part on a species by species basis, the task of identifying the various values and successes of a reintroduction project has important ramifications for determining the potential merit and effectiveness of reintroduction programs in general.

Since 1973, the endangered red wolf (*Canis rufus*) has been the focus of a federal recovery program (Carley 1975, U.S. Fish and Wildlife Service 1984, Parker 1988) (see case history). In 1987, the U.S. Fish and Wildlife Service (USFWS) intensified recovery efforts by initiating a reintroduction project at the Alligator River National Wildlife Refuge (ARNWR) in northeastern North Carolina (Smith and Phillips 1987, Phillips 1988, Phillips and Parker 1988, Meese 1989, Phillips 1990, Parker and Phillips in press). In this paper I discuss some interim measures of the value and success of this ongoing project.

Measures of Value

The reintroduction of red wolves into ARNWR was accomplished only after the USFWS carried out an education program that prompted many people to change their attitudes toward other species, even so-called "varmits" like wolves. The change represented a shift away from the historic belief that wolves are a serious and consistent threat to human safety and a competitor with hunters for game. Since the first red wolves were released, the project has been offered as badly needed proof that wilderness species and humans can coexist.

Red wolf (*Canis rufus*) Photo by M.K. Phillips

"Intensive management programs, like the one developed for red wolves in the refuge, will be necessary if many endangered species are to persist and evolve."

The red wolf is believed to have evolved solely in North America (Nowak 1979). Thus, the species is an important part of the history and heritage of the United States. However, prior to reintroduction at ARNWR most U.S. citizens knew very little about red wolves. The opportunity to learn about free-ranging red wolves might have been forever lost if not for the ARNWR reintroduction project.

The reintroduction project allowed the USFWS to develop the red wolf into an effective "flagship" species for conservation. Quite honestly, the task was easy because wolves evoke strong emotions in people. Regardless of whether people are for or against wolf reintroduction, most are interested in the project. Thus, since 1986, the USFWS has been able to use the ARNWR red wolf project as a vehicle to present information not only about wolf restoration, but also about the plight of other endangered species and environmental issues.

The red wolf reintroduction program also portends the future for many species. As humankind continues to modify the landscape, animal and plant species will be squeezed into smaller and smaller islands of suitable habitat. Intensive management programs, like the one developed for red wolves in the refuge, will be necessary if many endangered species are to persist and evolve. The ARNWR project provides conservationists with the opportunity to study and begin to perfect the process of ecological restoration.

The annual budget for the ARNWR reintroduction project is about $160,000.

Since most of this money is spent in northeastern North Carolina, the project provides monetary benefits to citizens of this area. However, in addition to generating direct monetary benefits, the reintroduction has generated a great deal of free publicity for Dare County (where ARNWR is located). Since 1986, a minimum of 22 magazines and 24 newspapers published stories about the project. Regional newspapers repeatedly covered the project. The project was discussed during the nightly newscasts of five national and four regional television networks; WVEC and WTKR, both based in Norfolk, VA, and both with access to very large markets, repeatedly covered the project. Three mini-documentaries were produced, including one by the Australian Broadcasting Company, and the red wolf was featured in the nationally broadcast "World of Audubon" documentary about restoration. Additionally, local radio stations presented information about the reintroduction. All stories about the ARNWR red wolf project depicted Dare County as an area that has escaped the trappings of the 20th century and whose natural resources are still healthy and thriving — the same image which local businesses and politicians portray in their advertising. Thus, the reintroduction project also indirectly benefits the local economy, since the fiscal health of the County depends almost solely on tourism.

Media coverage effectively informed millions of people about the reintroduction project. Thirty-three people became so committed to the concept of restoration that they donated approximately 10,000 ha to the project. In addition, many volunteers were utilized, most of whom were recent college graduates or students completing degrees in wildlife management. All volunteers received extensive training in red wolf restoration and eventually made significant contributions. The tremendous opportunities afforded young biologists in need of experience is one of the most striking examples of the project's value.

In addition to stimulating individual action, the reintroduction project prompted civic groups and private companies to become involved with conservation. For example, the North Banks Rotary Club (Kill Devil Hills, NC) developed a "conservation internship program" that consisted of a weekly stipend of $50 to $100 provided to individuals volunteering for the project. The program made a significant contribution to red wolf restoration, shielded the volunteer program from the vagaries of federal funding, and provided opportunities to people interested in conservation.

The red wolf reintroduction project also prompted the Conservation Fund (Washington, DC) to acquire approximately 47,000 ha of coastal plain habitat west of ARNWR. The acquisition will be managed as a conservation area by the USFWS and the North Carolina Wildlife Resources Commission. This additional acreage secures critical habitat for countless wildlife species in addition to red wolves, and provides significant protection to the Alligator River watershed and associated Albermarle-Pamlico estuarine system.

"Producing and raising offspring in the wild is irrefutable evidence that red wolves can make the transition from dependency on humans for food to self-sufficiency. However, the presence of wild-born wolves is just one component of a successful [reintroduction] program."

Evidence of Success

Since the ARNWR red wolf reintroduction was a first, there was no accepted definition of success against which to compare the project's progress. The technical proposal developed for the project defined success as the presence of second generation wild-born pups in the refuge (Parker 1986) — a definition developed mostly to provide USFWS officials a yardstick with which to measure the project's progress. Pups were born to free-ranging wolves during spring 1988. In fact, four pairs of wolves produced litters, of which pups from each are still alive (Phillips 1989). Producing and raising offspring in the wild is irrefutable evidence that red wolves can make the transition from dependency on humans for food to self-sufficiency. However, the presence of wild-born wolves is just one component of a successful program.

Another measure of success is the biological information gained through associated research and monitoring of the project. The backbone of the project consists of radio-tracking released individuals to monitor the results of reintroduction efforts. To date, over 4,000 relocations have been recorded (U.S. Fish and Wildlife Service unpubl. data), providing information about red wolf home range characteristics, food habitats, activity patterns, sociality, reproduction, and mortality. This information is available to interested individuals simply by contacting the refuge office. This increase in knowledge about red wolves and its accessibility to interested members of the public is another indicator of the program's success.

Although the first three years of the project presented some very difficult management situations, the monitoring program allowed the USFWS to stay only a few steps behind the problems. Since the first wolf was released, 17 animals had to be recaptured on 26 occasions. In spite of our preparedness, recaptures took place under conditions that were usually less than ideal. Nonetheless, recaptures were carried out without inflicting significant long-term damage to animals and with little inconvenience to residents of the area. Successfully managing the wolves helped to convince USFWS officials and local citizens that wolf restoration can be carried out in a controlled manner. This "track record" developed at ARNWR will be a tremendous aid to the

USFWS as it prepares and implements wolf restoration programs elsewhere.

On the other hand, 15 of the 29 released wolves died during the first three years of the project. To some, 50% mortality is unacceptable and evidence that the program is a failure. The USFWS feels, however, that 15 deaths are not excessive. In fact, the USFWS believes it is a measure of the program's success that all deaths were natural or accidental, and not the result of a citizen acting irresponsibly or on some unfounded hatred for wolves.

Assessing the value and success of an endangered species reintroduction program is not an easy task. For the ARNWR red wolf project, measures of the value and success of the reintroduction are varied. Some are obvious and easily defined quantitatively, whereas others are subtle and not definable in monetary terms. Nonetheless, each measure of value and success provides justification and evidence of the manageability of this landmark restoration project upon which the very existence of a species may depend.

The values and successes of reintroduction projects often have the potential to extend beyond the immediate preservation of the reintroduced species, to positively affect local citizens and communities, larger conservation efforts, and other imperiled species as well. Determination of the wide range of possible benefits and successes of a project may be useful in weighing the desirability of other captive breeding and reintroduction initiatives in the future.

"The values and successes of reintroduction projects often have the potential to extend beyond the immediate preservation of the reintroduced species, to positively affect local citizens and communities, larger conservation efforts, and other imperiled species as well."

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Female red wolf

Photo by M. Phillip

Ethical Perspectives on Captive Breeding: Is It For the Birds?

by
Gary E. Varner and Martha C. Monroe

Introduction

On the surface of it, what could be more ethical than struggling to preserve an endangered species through a captive breeding program? The dedication of the many individuals who have sacrificed to establish captive propagation facilities across the country, and the thousands of volunteers who help make them work, is undoubtedly laudable. However, a closer look at the various ethical justifications which might be offered for captive breeding reveals a more complex picture.

Our thesis is that, given the limitations of captive breeding technology and the financial limitations of the species preservation movement, today's captive breeding programs can only be viewed in a strongly positive light from a very narrow anthropocentric perspective, a perspective which matches neither the rhetoric nor the intuitions of those in the forefront of the environmental movement. We do not mean to suggest that all captive propagation ought to be abandoned, but we do argue that sometimes, when a choice must be made between saving a particular species through captive breeding and acquiring important habitat for in situ preservation of a whole biotic community, we ought to choose the latter and knowingly consign a species to extinction. Such choices will be difficult, but our examination of the ethics of captive breeding suggests that current priorities need some rearranging.

Three Ethical Perspectives

Captive breeding programs can be evaluated from three very different ethical perspectives. [Other perspectives are possible, but these three represent those most commonly encountered in the literature of environmental ethics and the rhetoric of environmental politics.] Someone who argues that endangered species should be preserved because they contribute to the welfare of present and future generations of human beings would evaluate programs from an anthropocentric perspective. Philosopher John Passmore (1974) adopted this perspective in *Man's Responsibility for Nature*, the first book-length treatment of environmental ethics by a professional philosopher, as did economist William Baxter (1974) in his widely-cited book, *People or Penguins: The Case for Optimal Pollution*. From these authors' purely anthropocentric perspective, endangered species should be preserved only if and insofar as doing so is justified on economic, scientific, recreational, religious, or other grounds tied directly to human interests.

". . . today's captive breeding programs can only be viewed in a strongly positive light from a very narrow anthropocentric perspective, a perspective which matches neither the rhetoric nor the intuitions of those in the forefront of the environmental movement."

A very different perspective, and one espoused by many environmentalist, is the holistic perspective championed by Aldo Leopold in *A Sand County Almanac*. In his summary statement of the "land ethic," Leopold defined morally correct actions as those which "tend to preserve the integrity, stability, and beauty of the biotic community" (Leopold 1948, pp. 224-225). Leopold's principle is holistic, because it subordinates the interests of individual members of the biotic community — be they deer or human beings — to the welfare of the community as a whole (Callicott 1989). Those who argue that all wild species ought to be preserved, even if doing so would not benefit human beings, usually are arguing from such a holistic stance.

A third ethical perspective frequently is called the "animal rights" position, but this label is misleading. So-called "animal rights advocates" usually do not defend the rights of all animals, but only "higher" animals like mammals and birds. This is because, from the animal rights perspective, what matters is the welfare of "sentient" beings (that is, conscious, experiencing beings), and presumably many "lower" animals are not sentient. [The "animal rights" label is also misleading insofar as many of those who argue from this perspective (including Peter Singer) are not advocating what a philosopher would recognize as rights for non-human animals. See Singer 1978, pp. 119-25, and Regan 1983, sections 6.4 and 6.5.]. The most widely-read account of the sentientist perspective is Peter Singer's *Animal Liberation* (1975), but the most philosophically rigorous is Tom Regan's *The Case for Animal Rights* (1983). Although Singer's and Regan's accounts differ greatly in detail, they are both (as we will put it) sentientist. Sentientists share the anthropocentrists' emphasis on individuals, rather than ecological systems. Both find intrinsic value in individual organisms, and only instrumental value in ecological systems. Sentientists simply find intrinsic value in a wider range of individuals.

In practice, environmentalists commonly argue from a complex mixture of these three ethical perspectives. Although anthropocentric considerations predominate in the policy arena, environmentalists commonly claim that environmental problems will not ultimately be solved until we adopt and act on a non-anthropocentric ethic. For instance, Denis Hayes, national coordinator of the first Earth Day, described the goal of the event in these terms: "We hoped it would lead to a new kind of ideology, a new value system based on ecology and a reverence for [all] life" (New York Times, 4/16/90). Similarly, in 1980, the United Nation's World Conservation Strategy declared that:

> "*Ultimately the behavior of entire societies towards the biosphere must be transformed if the achievement of conservation objectives is to be assured. A new ethic, embracing plants and animals as well as people, is required for human societies to live in harmony with the natural world on which they depend for survival and well-being* " (World Conservation Strategy 1980).

In such statements, the sentientist and holistic perspectives tend to become blurred together, but what is clear is that environmentalists commonly place a heavy emphasis on non-anthropocentric ethical perspectives.

Evaluating Captive Breeding

But how do captive breeding programs look from these non-anthropocentric perspectives? From the perspective of individual sentient animals involved in a program, captive breeding is a moral atrocity. Economic constraints prevent confinement systems from allowing birds and mammals a degree of freedom of movement comparable to the wild — a condition which must be especially distressing to first generation breeders who have known freedom. Also, very invasive techniques must be used to maintain full genetic representation and prevent inbreeding depression, including euthanization of offspring with overrepresented genes, embryo transfers between species, injections to superovulate females, and double-clutching.

From a sentientist perspective, even if a species is going extinct in the wild so that captive breeding is the only possible way to preserve it, it is still difficult or impossible to justify captively breeding the remaining individuals. Singer and Regan both acknowledge this, and Regan labels the differential treatment of members of endangered species "environmental fascism" (Regan 1979, sec. 9.3; see also Singer 1979). However "good" a program may be when seen from the perspective of the species, a captive breeding program cannot plausibly be said to be good when viewed from the perspective of the individual animals involved. Where the individuals involved are not sentient (plants, for instance, or mollusks or insects), a program is not sullied from the sentientist perspective. However, to the extent that economic limitations force us to focus resources primarily on the higher avian and mammalian taxa, captive breeding programs can receive little or no support from the sentientist perspective.

". . . from the holistic perspective, today's captive breeding programs are a rear-guard action in a war which we are still losing."

From the holistic perspective, these same resource limitations markedly limit the value of captive breeding. The holistic perspective places a premium on preserving the integrity of natural ecosystems. But it is impossible to preserve the integrity of an ecosystem without preserving all (or at least most) of its component species. To the degree that the holistic perspective is truly ecosystemic, it will find relatively little value in programs which, like today's captive breeding programs, tend to preserve only the large, impressive mammalian and avian species and consigns the far more numerous endangered species of mollusks, insects, plants, and fungi to extinction.

The value of captive breeding programs is further limited, from the holistic perspective, by the degree to which successful reintroduction is thwarted by continuing habitat destruction and the maladaptive influences of captivity. If the success of a captive breeding program is measured in terms of the likelihood of eventually restoring the organisms to a natural system, then captive breeding must be seen as a way to buy time for effective habitat preservation and enhancement. From the holistic perspective, this must be at the core of a successful program. A related problem is that prolonged captivity tends to erase learned information about feeding and nesting sites and learned behaviors related to reproduction. It is also a concern that after several generations in captivity, genetic selection inevitably begins to favor traits suited to captivity — traits which are unlikely to be adaptive upon return to the wild.

"... when a choice must be made between saving a particular species through captive breeding and acquiring important habitat for <u>in situ</u> preservation of a whole biotic community, we ought to choose the latter and knowingly consign a species to extinction."

All of this suggests that only when captive breeding is complemented by an aggressive program of habitat preservation does it look good from the holistic perspective. Yet, the primary cause of species endangerment still remains habitat loss. So from the holistic perspective, today's captive breeding programs are a rear-guard action in a war which we are still losing.

Hence, it is only from the human-centered perspective of anthropocentrism that great value can be placed on today's captive breeding programs. Even here, however, it makes a great deal of difference which human interests captive breeding programs are expected to serve.

If our only goal is to preserve unusual strains of food crops or medicinal plants as genetic resources, then although resource constraints severely restrict the number of species that can be thus preserved, captive breeding can be seen as a highly valuable technology. As long as we have no interest in in situ preservation, captive breeding looks good from an anthropocentric perspective.

However, from a more enlightened anthropocentric perspective, captive breeding programs score poorly for the same reasons they scored poorly from the holistic perspective. In *Why Preserve Natural Variety?*, philosopher Bryan Norton (1987) defends such an enlightened anthropocentrism. He argues that, even if only one in 10,000 species has direct value to human beings, all species have indirect value, because diversity begets diversity: increasing diversity increases opportunities for niche specialization, which in turn produces more diversity, and so on, in an ascending spiral which produces those relatively rare species that are of direct use to us. And in a period of rapid biological impoverishment, each species' value increases dramatically, because loss of diversity similarly begets further loss of diversity in a descending spiral. Thus, an enlightened understanding of our interest in preserving species as genetic resources would lead us to preserve evolving ecosystems. But as we have already seen, from such an ecosystemic perspective, a captive breeding program can be only as successful as a concurrent program of aggressive habitat preservation and enhancement. Enlightened anthropocentrists find instrumental value in ecosystems, whereas holists find intrinsic value there, but neither perspective can endorse captive breeding without habitat preservation.

"Enlightened anthropocentrists find instrumental value in ecosystems, whereas holists find intrinsic value there, but neither perspective can endorse captive breeding without habitat preservation."

Conclusion

We conclude that, given the limitations of captive breeding technology, and given the financial limitations of the current species preservation movement, captive breeding programs can only be viewed in a strongly positive light from a very narrow anthropocentric perspective emphasizing species' resource value to present and future human beings. Without meaning to trivialize this value, we emphasize that it does not represent the kind of non-anthropocentric perspective that so many environmentalists believe should guide our thinking.

How should environmentalists resolve this impasse? One way would be to abandon their non-anthropocentric rhetoric. Yet, as Norton's argument shows, even the more enlightened anthropocentrists cannot support captive breeding without effective habitat preservation. The upshot is that both environmentalists arguing from a holistic perspective and conservationists arguing from an enlightened anthropocentric perspective should push for an intelligent balance between habitat preservation and captive breeding. While sometimes the preservation of a particular endangered species can be accomplished through habitat preservation, all too often it is too late for this strategy to work. These are the cases in which a balance must be struck. Where resources are scarce, it cannot plausibly be claimed that captive breeding is always warranted. For instance, it is not clear that we should spend millions of dollars to breed a species like the California condor, which has a thriving functional correlate (the turkey vulture), if the money could be redirected to habitat acquisition which would benefit other species in situ. "Flagship" projects, like the California condor recovery program, may ultimately generate a great deal of public support, but if we want to be true to the holistic ideals of the environmental movement, we must openly question the wisdom of captive breeding in such cases.

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Captive Propagation and the Conservation of Species: A U.S. Fish and Wildlife Service Perspective

by
James Tate, Jr.

Intent of the Endangered Species Act

In writing the Endangered Species Act (the Act), Congress specifically stated that the Act's purpose is to conserve the ecosystems upon which endangered and threatened species depend. This concept — that it is preferable to conserve species by conserving ecosystems — underlies the entire Act. Yet in the publicity that often attends last-ditch efforts to save a species, the central importance of ecosystem conservation is often forgotten. What is left in the mind of the public is the misleading and dangerous idea that the Act's purpose might be to save species one by one, in isolation. And possibly worse, that the success of the Act should be measured by the success of such efforts.

Even though the U.S. Fish and Wildlife Service (USFWS) conducts research on ways to raise plants and animals in captivity and release them into suitable habitat, the USFWS uses captive propagation and release only as a technique of last resort for recovering a species from the brink of extinction. Examples of species for which the USFWS is attempting this approach include Knowlton's cactus (*Pediocactus knowltonii*), Kearney's bluestar (*Amsonia kearneyana*), desert fishes, peregrine falcon (*Falco peregrinus*), California condor (*Gymnoglyps californianus*), and black-footed ferret (*Mustela nigripes*) (see related case studies). The use of captive propagation with respect to any species depends on many things, including the feasibility of other less costly, or more effective approaches. But fundamental to implementing the Endangered Species Act is an understanding of this fact: maintaining species only in captivity actually detracts from the Act's basic goal of conserving functioning ecosystems.

> **"Congress specifically stated that the [Endangered Species] Act's purpose is to conserve the ecosystems upon which endangered and threatened species depend."**

Success Stories

It is, of course, true that some species can be raised in captivity and returned successfully to the wild. The best example may be the success of the captive rearing and reintroduction program for the peregrine falcon. This cooperative effort of the public and private sectors of the conservation community was made possible in part by the fact that nearly five centuries of recorded history exist on techniques for returning hand-reared birds to the wild after a period of loose control by humans. This process (called "hacking"), although developed over centuries by trial and error, has only recently been used successfully and systematically to return such species as peregrine falcons and bald eagles (*Haliaeetus leucocephalus*) to the wild.

Successful reintroduction of these avian species, however, was possible only because other major threats to their survival were being addressed at the same time. It was a combination of the United States' ban on chlorinated hydrocarbon-based pesticides, a concerted captive-rearing effort, and the application of hacking techniques that allowed peregrine falcons and bald eagles to increase. These efforts have been successful enough that it may even be possible to reclassify these species before long from "endangered" to "threatened." However, even reclassification will be controversial unless a vigilant watch is kept on other threats to the species' survival.

Questionable Programs

Unfortunately, for most species, such a combined storehouse of historical experience and environmental knowledge simply does not exist. For example, captive propagation "headstarting" efforts being undertaken by several government agencies for the recovery of sea turtles, in particular Kemp's ridley sea turtle (*Lepidochelys kempii*), do not appear to have materially advanced the recovery of these species.

The original agreement for this headstarting program was among the USFWS, National Park Service (NPS), National Marine Fisheries Service (NMFS), the Texas Parks and Wildlife Department, and Mexico's Secretary of Fisheries, with the USFWS serving as the coordinating agency. [This agreement involved far more initiatives than the headstarting experiment, and although not discussed here, success of other efforts must not be overlooked.] In the headstarting effort, the USFWS in cooperation with Mexico, moved approximately 2,000 Kemp's ridley eggs annually during 1978-1988 from Rancho Nuevo, Mexico (the only site where they nest), to the NPS's Padre Island National Seashore. They were turned over to NPS personnel who incubated the eggs until they hatched. The hatchlings were released briefly on Padre Island in an attempt to imprint the baby turtles on the beach and adjacent waters. They were then transferred to NMFS facilities in Galveston, Texas, where they were "headstarted" (maintained in tanks) for a period of months. In theory, the headstarted turtles then would be large enough to avoid as high a mortality rate as the vulnerable hatchlings experience. The headstarted turtles were released into the Gulf of Mexico, usually in the waters off Padre Island. The original agreement among all the parties involved was to conduct the Padre Island experiment for ten years. It was extended to 11 years, then terminated after the 1988 transfer of 1,000 eggs to Padre Island. [Under a separate five-year agreement, 2,000 Kemp's ridley hatchlings are now being moved annually from Rancho Nuevo to the NMFS facilities at Galveston for continuation of experimental headstarting.]

The USFWS continues to cooperate fully in the effort to conserve sea turtles, but it no longer advocates the headstarting project. In his testimony before the House Merchant Marine and Fisheries Committee, Conley Moffett, Deputy Assistant Director for Fish and Wildlife Enhancement for the USFWS, stated that "... at this time, with the data available, the Service does not endorse headstarting or any form of hatcheries for raising and releasing sea turtles into the wild as a recovery action or an attempt to maintain present numbers of reproducing marine turtles in the wild." He made it clear that the recovery of Kemp's ridley sea turtle has not been materially advanced by the program.

One would think that such an effort — by the agencies of two nations, private conservation organizations, and hundreds of volunteers — should make a difference. It would appear to be a sound conservation measure. But the bad news is that there is no evidence that headstarted sea turtles can be imprinted successfully to a natal beach. If everything worked according to theory, the turtles would return to the beach from which they were hatched when they reach maturity. So far, however, no one knows what cues it takes or for how long a sea turtle must be imprinted in order to provide the information it needs to return many years later to a specific site.

Like hacking hawks to the wild, headstarting turtles is not new. But it has been attempted in many areas around the world only for a few decades, with one of the largest and longest programs carried out by the State of Florida for over a 30-year period. But Florida's program, like almost every other effort, has been suspended. There has been no evidence that the state's long-term efforts, "or any other headstarting efforts, have resulted in a single headstarted animal surviving and entering a breeding population," according to Jack Woody, the USFWS's sea turtle coordinator.

Need To Pursue Other Strategies

Like many other species we are studying, relatively little is known of the life history or ecology of any of the several species of sea turtles. Although we are learning rapidly, the need is to find recovery answers. Several avenues usually need to be followed before we go to the ultimate, last-ditch stand of captive propagation. Research needs to continue on captive propagation, but not to the exclusion of other basic studies.

The example of the American alligator (*Alligator mississippiensis*) shows that captive propagation may not be necessary for recovery. This species has been brought back from the brink of extinction. Under strict state oversight, alligators are now actually being commercialized in certain areas. What the alligator needed for recovery was the same thing the sea turtles need: strict protection of nesting sites, strict protection from killing by humans, and long-term management programs to prevent extinction. With such protection, the USFWS believes the Kemp's ridley and other sea turtle species will begin returning to safer population levels. If this protection is not provided, then sea turtles, which have been around for 100 million years or so, may be lost to the world.

"... maintaining species only in captivity actually detracts from the Act's basic goal of conserving functioning ecosystems."

The hatchery concept has great appeal. It sounds ideal: if you need more turtles, raise them in a hatchery — just like trout, quail, pheasants, or parrots. However, the hatchery concept fails to address the problems that have brought many species to their present sad state. Until problems in their environment are recognized and corrected, we are doing little more than supporting a "put-and-take" operation. We are attempting to treat the signs of the problem, but not the actual cause of the problem. Unfortunately, there are no quick fixes for sea turtles or any other species on the brink of extinction.

Similar examples can be found in other hatchery-produced species. The introduction of non-native ring-necked pheasants to previously unoccupied habitat throughout central North America was successful only because of repeated introductions of previously wild birds to the new habitat. Most hatchery personnel assume that the vast majority of hatchery-reared birds released into a new area will be dead within a year. Pheasant populations that are constantly supplemented by hatchery-reared birds usually collapse soon after the cessation of releases. Similarly, hatchery officials know that only a tiny fraction of the millions of trout and salmon raised and released annually will survive to the point of reproducing successfully in the wild. It took even two tries to introduce the highly adaptable starling (*Sturnus vulgaris*) into North America, one hundred years ago this year.

Many other examples can be found of species not likely to benefit significantly from a propagation program, but which would benefit from other conservation measures, such as habitat conservation and control of take. Although in extreme situations it may well be necessary, the practice of raising species in captivity must not be allowed to overshadow the stated purpose of the Endangered Species Act: to conserve the ecosystems upon which endangered and threatened species depend.

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The California Condor: Current Efforts for its Recovery

by

Michael Wallace

Reasons for Decline

The California condor (*Gymnogyps californianus*) was probably never an abundant species. The fossil record indicates, however, that 11,000 years ago during the Pleistocene, the California condor once had an extensive range; it occupied the west coast extending from British Columbia to Baja California, as well as the southern-most states in the United States across to Florida, and even had a presence in upper New York state. Current hypotheses cite massive extinctions of the "mega fauna," the condor's food supply, during the latter Pleistocene as the reason for the species' reduction and restriction in range to only the West Coast (Emslie 1987).

Carrion found along the Pacific beaches provided by the still-abundant marine animal populations allowed the scavenger's survival into recent times. Over the last few centuries, loss of foraging habitat, encroachment on nesting territories, and direct mortality — such as collisions with human-made structures, ingestion of poison-contaminated carcasses, and shooting — are suspected causes of the condor's decline. By the early 1980s, only a remnant population of about two dozen birds remained.

Intensive field studies conducted by the U.S. Fish and Wildlife Service (USFWS) and others on the remnant population in the early 1980s involved an innovative photographic censusing technique and the marking of most individuals with wing tags and radio transmitters (Snyder and Snyder 1989, Snyder and Johnson 1985). During this period it was hoped that by quickly discovering the causes of the decline and immediately addressing the problems, the wild population could be saved and maintained in the natural environment with enhancement from young produced by a few captive pairs. Suspected causes of the decline prior to this study included lack of reproduction due to pesticide contamination or due to an insufficient food supply as a result of an obvious reduction in foraging area. While these factors may certainly have contributed to the precipitous decline, field studies conducted during the early 1980s indicated that reproduction was close to normal. Consequently, direct mortality alone could account for the population reduction.

An adult California condor at the breeding facilities in the Los Angeles Zoo Photo by M. Wallace

"...on April 19, 1987, the last bird was brought into captivity. All hope for the species' future rested with 27 individuals housed at two facilities..."

Captive Breeding

Condors are large, long-lived birds that reach sexual maturity only after five to six years of age. Their rate of reproduction is slow in the wild, with one chick being produced every two years. A species with these reproductive parameters must have a low mortality rate in order for the population to be somewhat stable; for condor populations to be maintained, annual mortality rates have been calculated to lie between five to seven percent (Verner 1978, Temple & Wallace 1988) Yet, from the early to mid 1980s, 23 to 40% annual mortality was documented in the wild condor population. This excessive and poorly understood rate of death eventually convinced the USFWS to capture the remaining wild condors for their own safety, and on April 19, 1987, the last bird was brought into captivity. All hope for the species' future rested with 27 individuals housed at two facilities: the Los Angeles Zoo and the San Diego Wild Animal Park. The program continues to rely entirely on captive propagation and the eventual release of genetically surplus young to the wild.

In 1987, the population consisted of 14 wild-caught juvenile and adult condors and 13 young that were artificially hatched from eggs taken from wild nests. The ability of female condors to lay a replacement egg if the first or even second egg is removed during the breeding season provides an opportunity to increase the reproductive rate up to six fold.

Compared to birds of prey, condors adapt to captivity with relative ease. The first signs of pair formation occurred in 1987, with two adults that had been in captivity for about one year. In 1988, this same pair laid one fertile egg that successfully hatched. In 1989, five pairs produced seven eggs, four of which were fertile and hatched healthy chicks, raising the population to 32 birds. In the 1990 season, nine pairs produced 15 eggs. Eleven of these were fertile and eight successfully hatched healthy individuals. As with several other species in captivity, eggs produced by first-time breeding condors, even if fertile, do not necessarily hatch, although their subsequent eggs have been viable. If all goes well, production may be as high as 13 or 14 young in the 1991 season.

Two events seem to be occurring reproductively with the captive flock at this point. Most of the wild-caught birds are now relaxed enough in captivity to breed, and the young of the "wild" eggs hatched in the early to mid '80s have now matured and show no reluctance to immediately begin breeding. Some have even shown reproductive activity as early as four and five years of age.

Release Criteria and Population Goals

If the current breeding success continues, releases to the wild can be expected to take place within the next few years. The timetable for future releases is set by criteria recommended by the California Condor Recovery Team, a panel of biologists which advises the USFWS on program direction. The four genetic and behavioral requirements for release candidates are:

1) at least 96% of the heterozygosity of each of the nine founding lines of the population must be retained in captivity before subsequent progeny can be considered for release (this amounts to five young per genetic line);

2) at least three pairs of founders are breeding;

3) candidates must be physically and behaviorally releasable; and

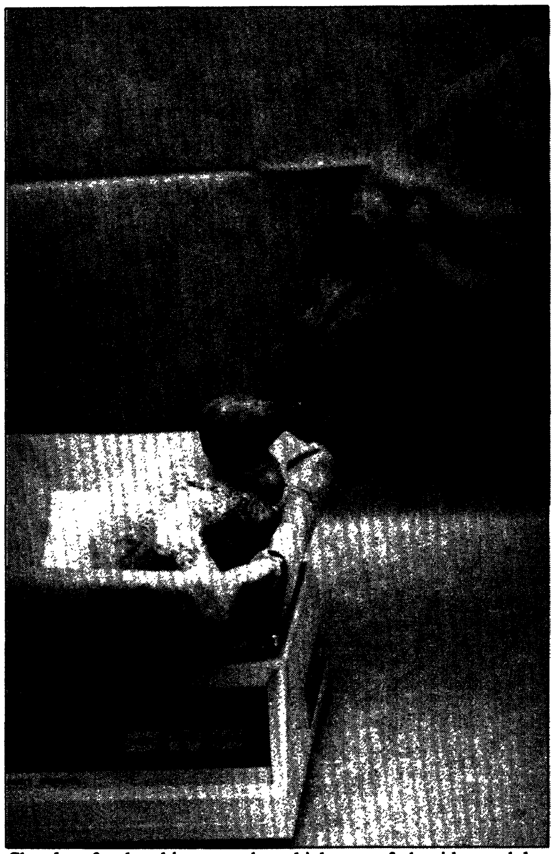
4) there must be at least three birds in any one release group.

Under these guidelines the reproductive pace of the birds themselves determine the timetable in which releases will occur. A few lines are close to meeting these criteria; if reproduction in the coming season is exceptional, releases could commence in 1991, however, it is more likely to occur in 1992. Although the first releases of California condors are scheduled to occur in sites tested in southern California, other areas both within and outside the state of California are being carefully considered. The list includes the Grand Canyon in Arizona and large tracts of land owned by The Nature Conservancy in southwest New Mexico.

The Recovery Team has recommended that the species be considered for downlisting to "threatened" when two wild, disjunct populations numbering at least 100 individuals each have been established.

Surrogate Work With Andean Condors

For many years, work with Andean condors (*Vultur gryphus*) has served as a model for the California condor program. Although endangered as well, populations of Andean condors found the length of the Andes mountains still number in the thousands. It is similar in size, behavior and ecology to its northern cousin. Because Andean and California condors are allopatric, lack of direct competition likely helped maintain these species as ecological equivalents with similar social, foraging and nesting characteristics (Brown and Amadon 1968). The close similarities between the two condor species have thus far ensured that techniques developed on Andean condors have been equally applicable for use on Californians.



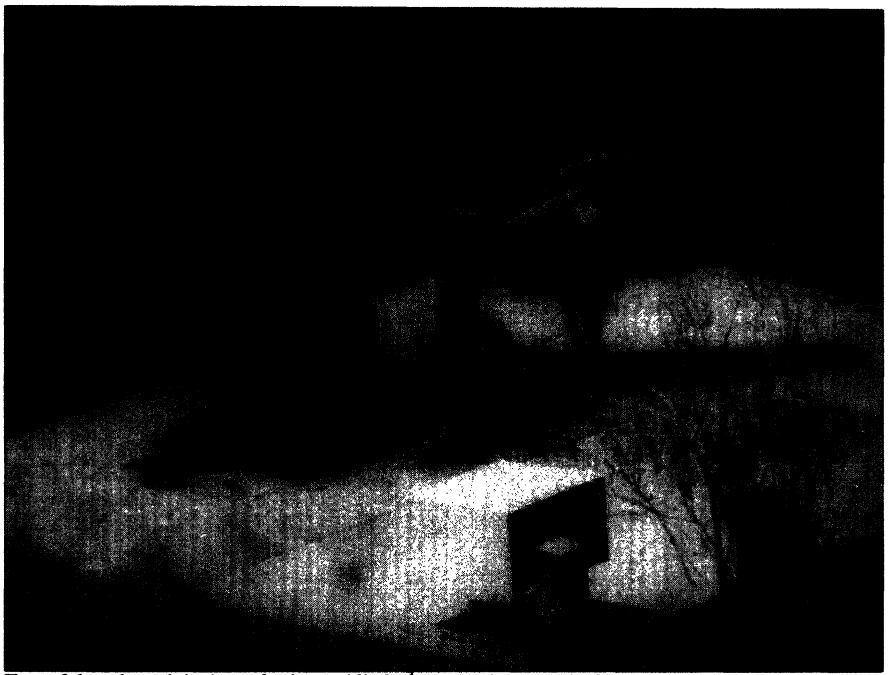
Shortly after hatching, condor chicks are fed with an adult-like puppet to orient them toward the correct species and avoid exposure to human contact. Photo by M. Wallace

Many zoos throughout the world have had success in breeding Andean condors (Cade 1986, Lint 1960). Artificial incubation and captive management techniques used on the California condors at the Los Angeles Zoo and the San Diego Wild Animal Park have been based on extensive experience accumulated by zoos working with eggs and young of Andean condors (Kuehler and Witman 1988). Radiotelemetry transmitters and tracking techniques were also developed with Andean condors, as well as trapping and field handling methods, prior to their use on California condors (Wallace and Temple 1987).

In anticipation of success with captive propagation of California condors, Andeans were used as surrogates to develop methods for releasing captive-raised condors to the wild. Based on successful releases of black vultures (*Coragyps atratus*) and turkey vultures (*Cathartes aura*) in Florida (Wallace and Temple 1983), 11 Andean condors were released in northern Peru. Seven survived and integrated successfully into the wild population over a two-year period (Wallace and Temple 1987).

The climate, topography, food dispersion and human activity levels are drastically different between the area that Andean condors were released in Peru and the southern California environment in which California condor releases are planned. While the captive California condor flock increases over the next few years to a level where young are available for release to the wild, Andean condors are again being used in the recovery effort to tailor condor release methods developed in Peru to the conditions that exist in the southern California mountains.

Since 1988, nine zoos have contributed 29 fertile Andean condor eggs to the



Two of the released Andeans feed on a 10' platform that keeps their food away from the competing black bears. Calf carcasses are acquired free from neighboring dairies. Photo by M. Wallace

program — their entire production for the 1988 and 1989 breeding seasons. Accompanied by zoo keepers, eggs were flown in portable incubators to the Los Angeles Zoo and San Diego Wild Animal Park for the 56 to 62-day incubation period. The chicks were reared in isolation from human contact using hand puppets modeled after adult condors. This strict isolation was maintained in the field release pens as well, when the nestlings were transferred at three to five months of age to the areas in which they would eventually be released. Maintained in groups of three to four at various sites, seven Andeans were released in 1988-89 and six in the 1989-90 season. These experimental releases were funded by the USFWS and the California Department of Fish and Game (CDFG), and continue to be jointly supervised by Los Angeles Zoo and USFWS personnel.

The importance of these "test runs" in release programs cannot be overly emphasized. The zoo production and management of the eggs and young in this program afforded the opportunity to assess the logistics of egg transport, improve incubation parameters, and revise chick rearing methods in view of later release results. Important refinements in condor release techniques are being made to make future releases of California condors more efficient and safe. For instance, we found that the release site must be carefully chosen by considering the management logistics of maintaining visual isolation while caring for the chick's physical needs, that the immediate topography must afford protection from large mammalian competitors and provide adequate ridge lift to encourage flight, and also that the degree of human activity in the surrounding landscape is an important factor.

Managing Causes of Mortality

In search of good flying conditions, birds in our first group were attracted soon after release to particularly windy slopes over a mile from the release area. These slopes were also heavily used by workers from the petroleum field located there. It appears that the close exposure to the high level of human activity, power lines and vehicular traffic over the first months of their fledging experience habituated the birds to those features in their environment. When they eventually ranged over a 100-mile area, they responded to the presence of people and human-made structures with far less fear than we would expect or like to see. Early in the releases, an inexperienced Andean condor died as the result of a collision with a power line in the same area on a prominent ridge. The incident underscored the necessity of selecting sites with a sufficient buffer of unobstructed slopes. The local oil companies that use the power lines have consented to bury the offending section and others similar in position to help prevent future incidents with California condors.

The remaining six condors of this first group were returned to captivity so they could not adversely influence six others subsequently released in a more isolated region of the range. As hoped, the second group chose a more secluded slope to practice soaring, and continued to respond much less positively than the first group to human activity when they later encountered it.

Lead poisoning by ingestion of bullet fragments from unretrieved deer carcasses during the hunting season has been identified as a mortality factor in the original free-flying population of California condors. A major question about the behavior of released condors is whether their feeding activity and foraging pattern could be controlled in a way that would minimize their exposure to this hazardous situation. By feeding the released Andeans on mountain peaks where soaring flight for the inexperienced birds is easy but where natural carcasses are scarce, the young condors have fed exclusively on food we offer, even though they traverse over 100 miles of wild habitat. Training the birds to a safer foraging pattern in this way is likely to help reduce mortalities in the first few years or possibly decades after California condor releases have begun, but is not likely to eliminate lead poisoning as a mortality factor for this species in the long term.

The best alternative to lead bullets being investigated at this time is copper. A pure copper slug in all the typical size loads for deer is currently being used in various parts of the country, and its manufacturer claims good reception by the hunting public because of superior ballistics and competitive price. If copper bullets prove non-toxic to the birds and their use can be encouraged in areas where condors and deer hunters coexist, then one major source of mortality can be reduced.

Andean condors may be used to teach the first release group of California condors a successful routine for life in the wild. If the current six Andeans are still free and behaving appropriately when California condors are ready for release, there is an opportunity to capitalize on the experience of the Andeans by having them show the foraging pattern, roost sites and drinking pools to the newly fledged Californians before the Andeans are again taken captive. Utilizing the experience of the Andeans in this way could significantly reduce the chances of the newly fledged Californians acquiring behaviors that are maladaptive, or at least, speed their adjustment to a wild environment.

Habitat Protection for Reintroduced Condors

One concern expressed by many people when all of the California condors were brought into captivity, was whether condor habitat would be lost to development and expansion of the human populace during the interim before condors could be released again into the wild. This concern begs the important question: exactly what is condor habitat? Their foraging habitat varies widely. One could safely say that it is wherever they can find carrion and are not disturbed while eating it. At one time this included the California beaches and the flatlands and open slopes around the San Fernando Valley — even the Los Angeles Basin itself. The foothills of the San Joaquin Valley have been most recently used extensively by the birds for foraging. Yet this area consists of a mosaic of privately owned land that is being continuously converted from ranching to agriculture at a rate that is unrelated to the condor's presence or absence. Condors will feed wherever food and opportunity are available — if establishment of safe feeding areas is deemed as a necessary management activity, then this will need to be done as necessary.

However, the "condor habitat" more critical to the future survival of the species is the nesting/living area found in the mountains of southern California. Fortunately, huge core mountainous areas currently are under protection by one governmental agency or another; the Sespe Condor Sanctuary, Hopper Mountain National Wildlife Refuge, and the Los Padres and Angeles National Forests represent relatively safe areas for condors to live. The agencies that preside over these important habitats, the USFWS, the CDFG, the Bureau of Land Management, the U.S. Forest Service and others, must be commended for continuing their protection of these lands even while lacking the presence of the birds.

"We are fortunate to have a test surrogate for the California condor such as the Andean [condor], but the methods developed using *Vultur* have also been applied to its own recovery as well."

Conclusion

We are fortunate to have a test surrogate for the California condor such as the Andean, but the methods developed using *Vultur* have also been applied to its own recovery as well. In order to ease concern over the possibility of Andeans proliferating in the wilds of North America, only females were allowed to be released. The 14 male Andeans that hatched during the California experimental program were also reared in isolation. However, they were then shipped to Columbia, South America, and released in areas were their species has declined or no longer exists. When the California experimental release program has been completed, the females will be flown to Columbia and released into the wild to join the males. Both condor species are directly benefitting from these programs.

If captive propagation of California condors succeeds to the degree we anticipate, adequate numbers of young should be available for annual releases. With basic release methods in hand, we are likely to witness significant increases in populations as we attempt to establish them in selected sites. Some mortalities will occur, but hopefully, with increased understanding learned with each circumstance, the causes can be dealt with on an individual basis. It appears that we can expect a reasonable amount of success in establishing populations — at least for the short term. What cannot be determined, at this point, is the long-term viability of a population recovering from such a tight genetic bottleneck. It will be many decades before we really know the success or failure of these efforts.

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The Decline and Restoration of the Guam Rail, *Rallus owstoni*

by
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INTRODUCTION

The Guam rail, *Rallus owstoni*, is a large rail (200 to 300 gms) known historically only from Guam, with no closely related forms nearby in Micronesia (Baker 1951, Ripley 1977). Baker (1951), however, speculates that the species may have had a wider range in former times. *R. owstoni* was distributed over much of Guam in all habitats except wetlands (Baker 1951, Beck and Savidge in press), although Jenkins (1979) considered both savanna and mature mixed forest to be marginal habitat. On Guam, the rail was an opportunistic omnivorous feeder that appeared to prefer animal over vegetable matter; it was known to eat gastropods, skinks, geckos, insects, and carrion, as well as seeds and palm leaves (Jenkins 1979).

The Guam rail was a year-round ground nester with a nesting peak occurring during the rainy season. Both sexes shared in the construction of a shallow nest made of loose grasses and in the incubation of eggs which lasted about 19 days (Jenkins 1979). The typical clutch consisted of three to four eggs, but the number of clutches per year on Guam was unknown (Jenkins 1979). The 172 broods observed in the wild ranged in size from one to four, with an average of two. One family group that was observed foraging in the wild consisted of two adults, several immatures and three chicks (R. E. Beck, Jr. and E. S. Morton, unpublished observations).

DECLINE OF THE GUAM RAIL

Prior to the 1960s, the Guam rail was abundant island-wide with its population estimated to be in the tens of thousands (Lint 1968). By 1982, a survey conducted by the Division of Aquatic and Wildlife Resources of the Department of Agriculture (DAWR), making use of tape recorded play-back of rail calls, found rails restricted to two small discontinuous populations in northern Guam (Aguon 1983). A repeat of the same survey in 1983 found an 80% decline from the previous year. By 1984, only approximately 20 rails remained in northern Guam, with the possibility of a few scattered isolated individuals elsewhere (Beck 1988a). Consequently, the Guam rail was listed on the U.S. Endangered Species List in 1984 (USFWS 1984a,b). The last confirmed observation of a Guam rail in the wild occurred in 1987 (Beck 1988a).

"The story of the decline of Guam's forest birds . . . is one of the most spectacular examples of the damage caused by an introduced species."

The story of the decline of Guam's forest birds (see particularly Savidge 1987) is one of the most spectacular examples of the damage caused by an introduced species (Pimm 1987). The spatial and temporal distribution of the introduced brown tree snake (*Boiga irregularis*) was well-correlated with the decline of Guam's native birds. Forest bird communities containing up to ten species lost all of those species, with population declines occurring sequentially in the central, southern, and northern portions of the island. When it became apparent in 1982 that many, if not all, of Guam's native forest birds would soon be extirpated, DAWR personnel made the decision to develop captive breeding programs for Guam's five remaining endemic species: the Guam rail, Guam Micronesian kingfisher (*Halcyon c. cinnomomina*), rufous-fronted fantail (*Rhipidura rufifrons uraniae*), Guam flycatcher (*Myiagra freycinen*), and bridled white-eye (*Zosterops c. conspicillatus*) (Beck 1988b). Unfortunately, the bridled white-eye, rufous-fronted fantail, and Guam flycatcher were so rare by mid-1983 that captive programs for these species had to be discontinued (Beck 1988b).

CAPTIVE BREEDING

Propagation efforts for the kingfisher and rail were subsequently accelerated, and a cooperative captive breeding program was organized under the auspices of the American Association of Zoological Parks and Aquariums (AAZPA). Initial participants included the DAWR, the Philadelphia Zoological Garden (Shelton 1986a,b,c), the Smithsonian Institution's National Zoological Park (Derrickson 1986a,b), and the New York Zoological Society's Bronx Zoo (Sheppard 1985). Between February 1983, and September 1986, 21 rails were taken from the wild as eggs, chicks, immatures, or adults for propagation purposes.

Because sexual maturity occurs at only four months of age and breeding is year-round, the captive population has grown rapidly despite pairing difficulties caused by high levels of aggression. The first successful breeding of the rail as a part of the captive breeding program occurred at the DAWR's facility on April 17, 1984 (Beck 1984). The first breeding of the rail at a mainland U.S. zoo occurred in September 1984, at the National Zoological Park (Derrickson 1986a,b). There are presently over 180 rails in captivity in 14 U.S. zoos and in Guam, derived from 13 of the original 21 potential founders. If three additional surviving wild-caught birds breed successfully, the number of founders can be raised to 16. Unfortunately, all attempts to pair and breed these three hand-raised birds have failed thus far due to social and sexual incompatibility. Fortunately, the captive population has continued to be characterized by an even sex ratio.

Although captive populations can obviously serve as important species reservoirs, they necessarily lack a number of important features of wild populations. Genetic variation can be quickly depleted in small populations as a result of genetic drift and inbreeding (Foose and Ballou 1988). Furthermore, behavioral adaptations necessary for survival in the wild may be progressively lost in the altered captive environment (Kleiman 1980). In short, re-establishing and maintaining a wild population should be a high priority for any captive breeding effort (Conway 1980).

Any reintroduction should attempt to provide as broad a cross-section of original genetic variation as possible in the released animals (Hedrick et al. 1986). Because the pedigree of the captive population of Guam rails is relatively shallow (Derrickson 1986a,b), current genetic and demographic management is aimed at maximizing founder genome equivalents — a strategy that effectively maximizes genetic diversity in both the captive and the reintroduced populations (Haig et al. 1990).

The potential loss of crucial behavioral adaptations is, of course, more difficult to predict. However, recent analyses involving birds and mammals (Griffith et al. 1989) and gamebirds (Witteman and Pimm in prep.) have demonstrated that reintroductions of wild-caught animals are much more likely to succeed than reintroductions of captive-raised animals. While a genetic explanation for these results cannot be ruled out, it seems likely that behavioral deficits in captive-bred animals are the likely cause of this difference. Although the captive breeding program for the rail has attempted to avoid the loss of wild behavior by maintaining birds in natural settings and minimizing inadvertent selection for tameness, progressive behavioral change through time in captivity is inevitable. Therefore, it is extremely prudent to proceed with re-establishing a wild population as soon as possible.

THE ROTA INTRODUCTION

Choice of Location

In planning the restoration of any species, one must answer the obvious questions of where the introduction should take place, how many individuals should be released, and how many releases will be required to establish a viable population. Because there is no known way to control or eradicate the brown tree snake on Guam at present (Savidge 1987, Fritts 1988), it was decided to establish the rails on other islands in the Marianas (Beck 1987, Beck and Savidge in press).

The island of Rota was subsequently selected for initial reintroductions for a variety of reasons. First, the island is located only 50 km north of Guam, and its proximity and access by commercial airline would minimize logistical problems. Second, the island has only about 1,500 inhabitants, which would allow for an effective public education program. Third, Rota's climate and plant and animal communities are nearly identical to Guam, and contains no predators not found on Guam. Finally, there are no other rails or ecologically similar species on Rota that might impact, or be adversely affected by, the introduction of the Guam rail (USFWS 1989). The presence or absence of ecologically similar species is known to influence the success of introductions on oceanic islands (Moulton and Pimm 1985).

Since detailed information on the rail's historical habitat preferences is lacking, we intend to release birds in a number of locations and habitats on Rota. By following the birds after release and determining their habitat preferences and survival, we will be able to better identify potential sites on the island for future releases.

Number of Releases and Birds Released

In the case of the Guam rail, there is necessarily a limit to the number of individuals that can be taken from the captive population for release. The current size and productivity of the captive population suggest that we will soon be able to produce between 75 to 90 rails per year for release, without jeopardizing either the genetic or demographic composition of the captive population. Yet even with this number of birds, many questions remain concerning the optimal number of birds per release, the frequency of releases, and the number of release locations. Fortunately, both

Guam rail (*Rallus owstoni*) Photo by Jessie Cohen, National Zoological Park theoretical insights and empirical studies can help us make these decisions.

Both environmental disturbances and demographic accidents appear to be important in determining the fate of small wild populations (Pimm et al. 1988). At extremely small numbers, breeding may produce only individuals of one sex, all the individuals may die in the same year from independent causes, or individuals may never even find a mate. These are demographic accidents, and by releasing more individuals, we minimize the chance that they will prevent successful introductions. Environmental disturbances (for instance, a major typhoon), result in many individuals dying of the same cause. As such, they can cause even very large populations to become extinct, or a very large introduction to fail. The obvious way to minimize such effects is to introduce individuals to different places. For instance, it is likely that some parts of the islands will fare better than others (in the example of the typhoon, some parts of the island may be better sheltered). This is a strategy of spreading risks, and it also helps to minimize the problems associated with our lack of detailed knowledge about the rail's habitat preferences and requirements.

The difficulty is obvious. With a fixed number of birds available for release, we cannot release large numbers of birds (to minimize the risk of demographic accidents), to large numbers of places (to minimize the risk of environmental disturbances). So what is the optimal solution and how do we find it?

We have used two approaches. The first (Pimm et al. 1988) examined the fate of several hundred small populations of about a hundred species of birds on islands off the British coast. Using an indirect and technical argument that space prevents our repeating here, we estimated that populations above seven breeding pairs were more likely to fail because of environmental disturbances than demographic accidents, and below seven pairs, the reverse was true.

The second approach examined the success of several hundred introductions of gamebirds into the United States (Witteman and Pimm in prep.). We considered only those species that were eventually successfully introduced. Interestingly, even for these species and even when large numbers of individuals were released, most introductions failed. This result suggests that the severity of environmental effects varies considerably, so choosing the right place and time for an introduction is important. On average, about seven releases were required before one was successful.

The relationship between success and the numbers of individuals introduced was logistic, that is, with increasing numbers, success first accelerated and then reached a plateau. The inflection point is where the per individual cost of becoming successfully established is minimized (around 75 individuals for the gamebirds), and the maximum success rate was about 15%. This relationship also suggests a strategy for optimizing the chance of a successful introduction. If several hundred individuals are available for release, several smaller releases in different places and at different times can have a higher probability of establishing at least one population. When the total number of individuals released falls below 75, splitting the release into smaller ones decreases the overall probability of success.

". . . progressive behavioral change through time in captivity is inevitable. Therefore, it is extremely prudent to proceed with re-establishing a wild population as soon as possible."

The optimal number for gamebird releases must be about 75 individuals, for reasons already explained. This is roughly comparable to the optimum result based on the island study, that is, seven breeding pairs. However, seven pairs (14 individuals) must be on the low side for an optimal release, for there would certainly be some non-breeding birds in the population. At the very least, these estimates (between 14 to 75 individuals) suggest that we have enough rails to attempt an introduction. If the optimal size for an introduction were an order of magnitude larger, for example, it would not be possible with the number of birds we now have.

In short, we estimated that the best way to introduce the Guam rail would be to split the large number of individuals available for release into smaller release groups, but not so small in size that the numbers fall to levels where demographic accidents are the principal cause of extinction. On the other hand, if only very few individuals were available, the release would have to be postponed until a larger number of individuals become available. The more releases attempted, the better the chance of hitting the right time and place. As indicated above, captive breeding efforts are such that we expect to be able to release 75 to 90 per year. This number provides enough individuals for several releases, and is close to the optimum release number providing that the birds remain near each other and form a breeding population.

First Rota Release

We released the first Guam rails on Rota between December 1989, and early February 1990. On Rota the birds were held for several days at a holding facility at the release site. During this time they were weighed, measured, and fitted with color leg bands and harness-mounted radio transmitters. The transmitters were of both solar and battery types and weighed between four to seven percent of the rails' body weight. The rails were fed a reduced portion of the feed used in the Guam breeding facility and a supplement of live-caught insects. Supplemental feeding began on Guam, and was intended to enhance the foraging abilities of the rails after release into the wild.

It became clear that use of radio transmitters would be critical in ascertaining the birds' disposition; because of their secretive behavior, there were only two confirmed sightings of rails one week after the release. Unfortunately, eight (36%) of the 22 birds released carried transmitters that failed, so we have no idea of their fate. Obtaining reliable transmitters is now a major concern. Of the birds we could track, there seemed to be little evidence of site fidelity, destroying any hopes that we could establish a viable population in the area we chose. The birds often moved along roads following their release, suggesting that they were seeking more open habitats than the one into which we released them. In choosing roads, the birds were vulnerable to traffic, and three birds were killed by vehicles. The other certain major cause of mortality was cat predation.

"The more releases attempted, the better the chance of hitting the right time and place."

Although the initial mortality was discouraging, it was by no means unexpected. Sources of mortality, such as vehicular kills and cat predation, can be prevented with appropriate measures. Currently, efforts to control feral cats are being made, and alternative sites are being considered that have less traffic. It is less obvious what measures are necessary to create a wild population in which encounters between individuals will be frequent enough for mating. Our plans call for additional releases of Guam rails on Rota over the next several years in both the dry and rainy seasons until a breeding population is established. The experience gained from these releases will accomplish several important goals: the Guam rail will have been returned to the wild at the earliest possible moment to a snake-free environment; a source of wild rails for possible future translocation to Guam will be established on Rota; and release techniques perfected on Rota will be available for future use on Guam when *Boiga irregularis* is controlled there.

Acknowledgements

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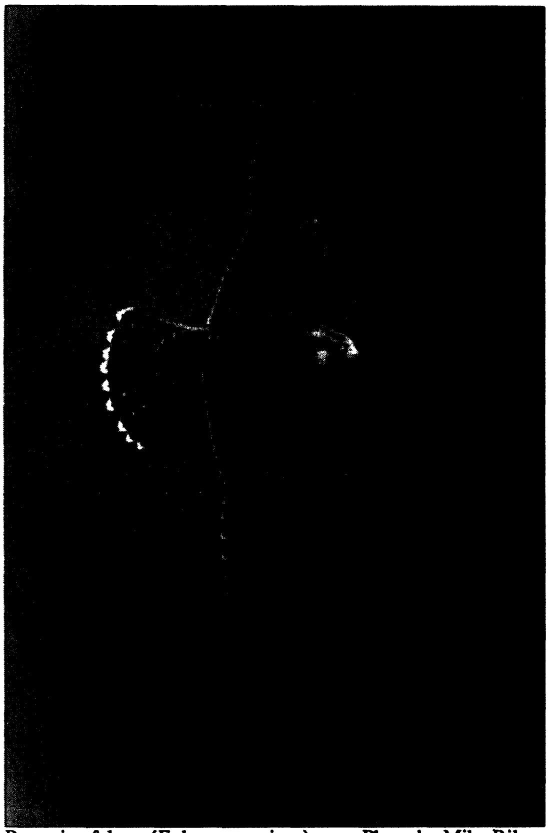
Peregrine Falcon Recovery

by
Tom J. Cade



Peregrine falcon (*Falco peregrinus*) Photo by Mike Riley

Species Decline

Most cosmopolitan and most admired of birds, the wide-ranging peregrine falcon (*Falco peregrinus*) experienced major losses in population during the 1950s, 1960s, and early 1970s, primarily in Eurasia and North America (Cade et al. 1988). Earlier there had been some local and regional reduction in the number of breeding pairs owing to climatic or other environmental influences, e.g., the loss of a sparse tree-nesting population along the Mississippi River prior to 1900 (Hickey 1942, 1969), the disappearance of peregrines and their replacement by prairie falcons (*Falco mexicanus*) in the intermountain region of the Pacific Northwest in association with a prolonged period of dry years beginning in the 1920s (Nelson 1969), and the loss of about 100 eyries in the western Scottish highlands owing to a reduced prey base associated with badly overgrazed sheep pastures (Ratcliffe 1980). It was, however, the heavy use — beginning in the late 1940s — of a new, potent class of chemicals, the organochlorine pesticides, that soon resulted in the unprecedented declines in peregrine populations recorded in the 1950s throughout Europe, the United States, and Canada (Hickey 1969, Cade et al. 1988).

The two main culprits were contaminants in the prey eaten by falcons: dieldrin (HEOD) acting directly to increase mortality, and DDT (DDE) acting indirectly through its effects on eggshells and consequent decrease in reproduction. The relative importance of these two factors apparently varied from region to region, and their respective roles in the continent-wide declines in Europe and North America have been much discussed (see various papers in Cade et al. 1988).

The end result of the combined effects of organochlorine toxicity was widespread abandonment of historical falcon eyries. In Europe, the loss of breeders ranged from complete extirpation for some regional populations (e.g., the tree-nesting peregrines in Germany, Poland, and the Baltic states), to reductions of 50 to 75% or more of original numbers in the British Isles and other parts of Europe.

In North America, nesting peregrines completely disappeared from the Mississippi River eastward to the Atlantic in the United States, and from most of contiguous southern Canada — in a region where an estimated 400 to 450 pairs formerly nested (Hickey 1942, Kiff 1988). (However, a single pair apparently held on right through the 1970s in southern Quebec.) By 1970, peregrines had also disappeared from the northern Great Plains of the Dakotas, Montana, and Alberta, where a sizeable population had nested on steep cut-banks along major rivers. A few pairs held onto their eyries and continued to breed at a greatly reduced rate in the Rocky Mountains of Colorado, southern Utah, western New Mexico, and northern Arizona, although they vanished from the northern Rockies. In California, numbers were reduced — especially along the coast — to about ten percent of the original population of some 200 or more pairs by the early 1960s. A comparable decline occurred in Oregon, again along the coast and along the Columbia River where 13 known eyries were abandoned. Events in Washington were not well chronicled except in the Okanagan Valley where nesting peregrines apparently vanished prior to the use of DDT (Nelson 1969). Further south in Baja California, peregrines disappeared entirely from the Pacific side, but they remained at their eyries on islands in the Gulf of California.

The Peale's peregrines (*F. p. pealei*) in the maritime habitats of the Pacific Northwest Coast from Washington north and west through the Aleutian Chain were never reduced much by pesticides except in Washington, but the decline which affected the continental peregrines eventually extended northward into the taiga and tundra regions of Canada and Alaska. By the early 1970s, local to regional declines amounting to 50 to 75% of the original number of breeders had been recorded in northern districts right across the continent (see various papers in Cade et al. 1988).

The low point in the number of breeders remaining in North America was reached around 1975 (Kiff 1988). It can be conservatively estimated at 30 to 50% of the pre-DDT population of approximately 10,000 adult pairs (see Figure 1).

Captive Breeding and Reintroduction Efforts

In response to these unprecedented population changes, the U.S. Fish and Wildlife Service (USFWS) first listed the *anatum* subspecies as endangered in 1969 under the old act; the *tundrius* subspecies, newly named by White (1968), was listed as endangered in 1970. The entire species has also been listed on Appendix I of CITES since 1975. In addition, use of DDT was greatly curtailed in the United States in 1972, followed soon afterward by dieldrin.

Interest in breeding peregrines in captivity as a tool for their preservation and reintroduction developed from the Madison Peregrine Conference in 1965, when it was first realized that the species was in serious trouble in both North America and Europe (Hickey 1969). A post-conference meeting of falconers and biologists resulted in the formation of the Raptor Research Foundation, Inc., as an umbrella organization to coordinate efforts to obtain suitable falcons for breeding, to establish bonafide breeding projects, and to disseminate information quickly on methods of breeding (Cade 1988). From the outset it was clear that this effort would involve both institutional programs and private projects. In the next ten years the following private breeders made important contributions to the development of breeding methodology in North America: Frank Beebe, Larry Schramm, Heinz Meng, James Enderson, John Campbell, R. Wayne Nelson, Robert Berry, and Lester Boyd. In the same period the following institutional operations got underway: the Canadian Wildlife Service facility at Fort Wainwright, Alberta, under the direction of Richard Fyfe; the USFWS's endangered species effort with beach-trapped peregrines at Patuxent, Maryland; and The Peregrine Fund, Inc., first at Cornell University under the direction of Tom Cade, later also at Fort Collins, Colorado, in cooperation with the Colorado Division of Wildlife, and finally permanently located at the World Center for Birds of Prey in Boise, Idaho, under the direction of William Burnham. Some other institutional programs joined in later: the Santa Cruz Predatory Bird Research Group at the University of California, Santa Cruz, under the direction of Brian Walton and closely associated with The Peregrine Fund; the Raptor Center at the University of Minnesota (Patrick Redig and Harrison Tordoff), the Macdonald Raptor Research Center of McGill University (David Bird), and the Saskatchewan Cooperative Falcon Project at the University of Saskatchewan, Saskatoon (Lynn Oliphant). In addition, two major programs also became established in Europe: one in West Germany under the direction of Christian Saar, in association with the University Falkenorden; and one in Sweden coordinated by Peter Lindberg at the University of Goteborg, in association with the Swedish Society for the Conservation of Nature.

Falcon breeders worked out the main techniques for propagating peregrines from 1965 to 1975 (Cade and Fyfe 1978, Cade 1980, Weaver and Cade 1983). It was learned, for example, that wild-caught immature or adult falcons could seldom be induced to breed in captivity, while hand-reared nestlings often could be, either by the mating of paired males and females or with the aid of artificial insemination, depending on how the falcons were treated during their formative period of socialization. Double and triple clutching or the sequential removal of single eggs proved to be effective methods for greatly increasing the productivity of captive females. Artificial incubation became the standard method for the development and hatching of eggs. A variety of feeding methods proved effective in rearing the chicks to maturity, but most commonly lab technicians feed them artificially by hand for the first ten to fourteen days, after which they are placed in the care of parent falcons for the remainder of their development and growth to flying age. About 90 to 95% of all chicks hatched can be successfully reared by this method.

By 1975, sufficient numbers of captively produced peregrines were being reared so that experimental releases could begin in order to learn the best methods for re-establishing these peregrines in the outdoors. That summer, The Peregrine Fund released 16 young by hacking at five sites in the eastern United States. Twleve of these birds were resighted at or near their release sites a year later (Cade and Dague 1975, 1976). The number of released falcons steadily increased to the current level of several hundred per year for all of North America, about 200 of which are released by The Peregrine Fund, Inc.

In all, more than 3,500 peregrines have now been released in the United

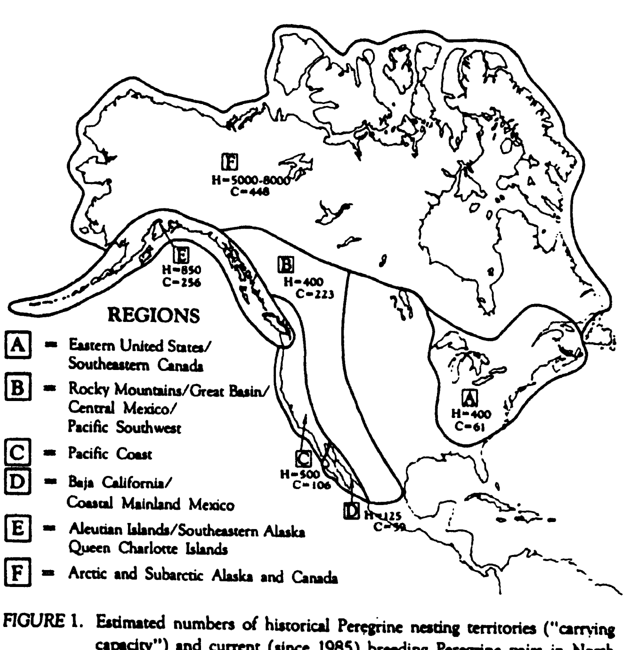


FIGURE 1. Estimated numbers of historical Peregrine nesting territories ("carrying capacity") and current (since 1985) breeding Peregrine pairs in North America. H=historical territories (n=7000-10,000); C=current confirmed pairs (n=1153+).

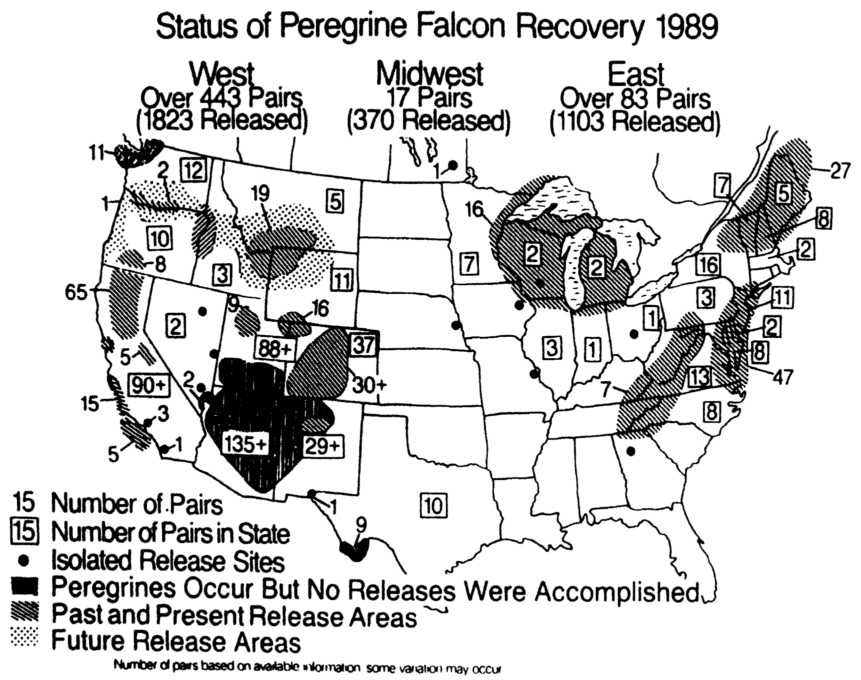
Reprinted with permission of Cade et al. 1988.

States and about 1,000 in Canada. Three methods have been used: hacking; fostering to wild peregrine parents; and, to a lesser extent, cross-fostering, mostly into the nests of prairie falcons (Sherrod et al. 1981, Barclay and Cade 1983, Cade et al. 1988, Linthicum 1989). Each method has proved to be about as equally effective as the others, although there are some reservations about cross-fostering.

Species Recovery

Peregrine populations in North America have been increasing dramatically since the late 1970s, both by natural processes and by artificial restoration (Cade et al. 1988). The efficacy of captive propagation and reintroduction must be evaluated within the overall context of this recovery (see Figure 2).

In North America, northern boreal and arctic nesting populations have shown the greatest natural increases since the low point reached around 1975. Some long-studied populations are now about twice as large as they were in former decades. For example, along the Colville River in Arctic Alaska where 32 to 36 pairs nested in the 1950s, now there are more than 50 pairs; along the upper Yukon in Alaska where 15 to 16 pairs nested in earlier years, there have been around 30 pairs in recent years (Ambrose et al. 1988); and the entire reach of the Yukon in Alaska had more than 125 pairs in 1990 (P. Bente, per. comm.). The population nesting around Sondre Stromfjord in western Greenland has steadily increased from five to six pairs found at 21 known eyries in 1974-1975, to 43 pairs at 67 cliffs in 1987 (Mattox and Seegar 1988), and the population has continued to increase since (T. Maechtle, per. comm.). Similar increases have occurred in Canada, and the overall increase in the breeding populations of boreal and arctic regions has been especially well-reflected by the number of fall migrants counted and trapped at Cedar Grove, Wisconsin, Assateague Island, Maryland/Virginia, and Padre Island, Texas (see relevant papers in Cade et al. 1988). These records indicate that the northern migratory populations nesting above 55 degrees N must be approaching 10,000 pairs.



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In the western United States, there are currently (1990) about 500 occupied eyries, but the rate at which new pairs are being found each year makes this figure very ephemeral. It is uncertain to what extent this number reflects the real increase as opposed to the discovery of long-used sites previously unknown. However, no one doubts that an increase has been underway for a decade or more. Released peregrines, identified by bands, occur at 100 to 150 of these eyries, mostly in California, Colorado, northern Utah, and the Greater Yellowstone Ecosystem, where they have contributed substantially to peregrine recovery. Indeed, there were no known breeding pairs in the Greater Yellowstone Ecosystem when The Peregrine Fund began reintroduction there in 1980. Yet by 1990, there were 28 occupied eyries (W. Heinrich, per. comm.), and it is likely that there are others to be discovered in this rugged, mountainous region.

In eastern North America south of the boreal forest, there are now more than 100 re-established pairs of peregrines in the Maritime Provinces of Canada, southern Quebec, and Ontario, in 11 states from Maine to South Carolina and in five mid-western states. As far as is known, these are all released falcons or their progeny, although the recent discovery of a substantial number of nesting peregrines extending down the Labrador coast suggests the likelihood that there will soon be an exchange of individuals between this wild population and the released falcons which have been expanding slowly up along the Gulf of St. Lawrence.

Uniqueness of Reintroduced Populations

Thus, approximately one-fourth of the original nesting population has been restored to this eastern region of North America through the use of captively produced peregrines for reintroduction. As Cade, Redig, and Tordoff (1989) have discussed, the restored population is not an exact ecological or genetic duplicate of the original. In particular, the reintroduced peregrines so far have been unable to reclaim the formerly used nesting habitat along the major, lowland rivers such as the Mississippi and Susquehana because of predation by great horned owls (*Bubo virginianus*). On the other hand, the released falcons have learned to use a variety of novel nesting biotypes that were seldom used by the former "duck hawks" — these include special nesting towers in coastal salt marshes, bridges, decommissioned ships, and skyscrapers in major cities.

Urban environments seem to be especially favored by released peregrines, and the number nesting in cities has increased rapidly (Cade and Bird in press). In 1990, at least the following cities had nesting or territorial pairs established: New York topped the list with nine to ten pairs, then Minneapolis-St. Paul with five pairs, followed by Los Angeles, San Francisco, Long Beach, San Diego, Nampa, Calgary, Edmonton, Winnipeg, Saskatoon, Ottawa, Quebec City, Montreal, Chicago, Milwaukee, Detroit, Toledo, Cleveland, Pittsburgh, Boston, Springfield, Philadelphia, Atlantic City, Baltimore, and Norfolk. Single birds occur in many other cities.

"... nowhere within its global range has the peregrine been able to re-establish breeding populations by natural processes (dispersal of surplus birds) where it was totally extirpated by pesticides in regions measuring from hundreds of thousands to millions of square kilometers ... It is precisely in such regions where captive breeding and reintroduction can play their most important role in species restoration."

Conclusion

Peregrines still remain absent from one large region of North America: the northern Great Plains of Montana and Alberta, where they once nested in considerable numbers on the cut-banks of major rivers such as the Missouri and Saskatchewan and their tributaries. It is interesting to note that nowhere within its global range has the peregrine been able to re-establish breeding populations by natural processes (dispersal of surplus birds) where it was totally extirpated by pesticides in regions measuring from hundreds of thousands to millions of square kilometers (e.g., eastern North America, the tree-nesting region of northern Germany, Poland, and the Baltic states). It is precisely in such regions where captive breeding and reintroduction can play their most important role in species restoration. The last major effort involving peregrines in North America should be a joint U.S.-Canadian program to repopulate the northern Great Plains.

There can be no doubt that restricting the use of organochlorine pesticides has been the most important action taken to benefit the recovery of this species — both in North America and in Europe. Captive breeding and reintroduction, however, have also played important roles in restoring populations in regions where peregrines were totally extirpated, and in speeding the recovery of remnant populations. Similar conservation initiatives can be widely applied to other birds of prey as needed, and indeed, to many other kinds of birds and animals as well.

As a consequence of the greatly improved status of populations, the USFWS downlisted the *tundrius* subspecies from endangered to threatened in 1984, and it should soon be a candidate for delisting. The time has also come to consider downlisting or delisting *anatum* populations on a regional basis in western states, particularly those in California, Arizona, New Mexico, Utah, and Colorado. It is indeed gratifying to all who have worked so diligently on behalf of peregrine recovery during the past 20 years to see the objective of the Endangered Species Act reaching fulfillment in the case of this species.

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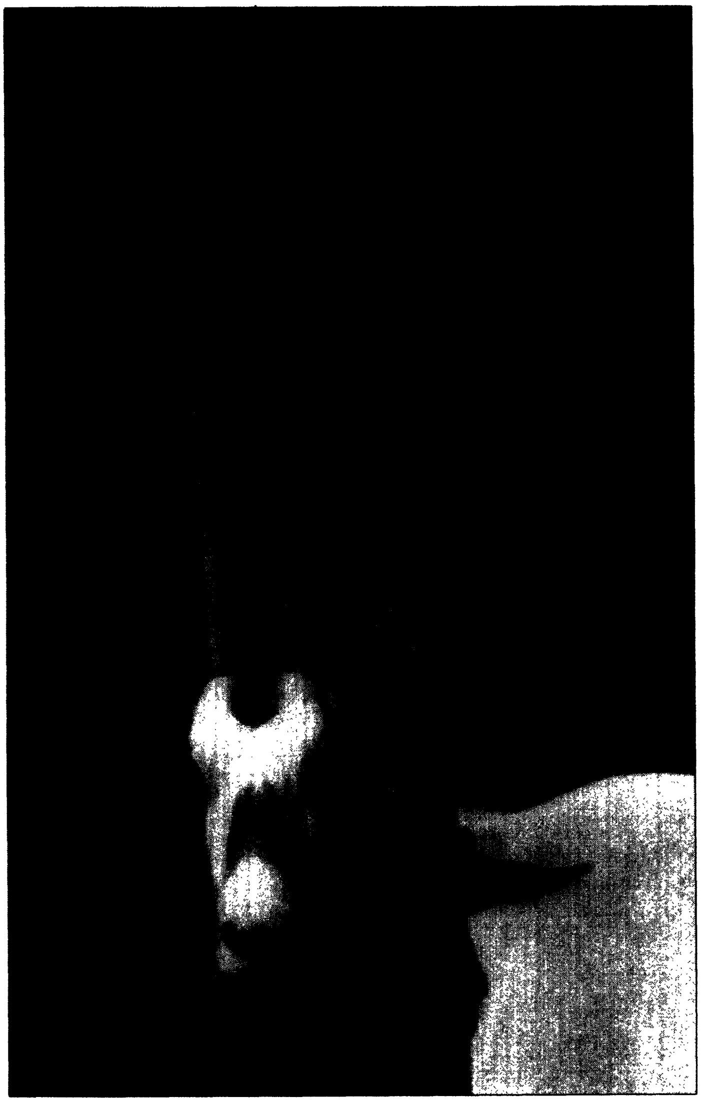
Captive Breeding and Reintroduction Programs

Nene or Hawaiian goose (*Nesochen sandvicensis*)
Photo by Jeff Black

This photographi is taken of a young male Hawaiian (Nene) goose in Haleakala National Park, recently fitted with a new plastic leg band — inscribed with the letters "AJ" — that can be read with a telescope from 250m. The bird, which is the progeny of released parents, was captured by State Forestry & Wildlife and National Park Service biologists in August as part of an increased initiative in the recovery program. The new initiative includes research and management programs to improve Nene habitat by enhancing feeding opportunities and reducing predators. The major thrust of the program, which includes a public information campaign, is due to begin in 1991 when initial funding is anticipated..
Contacts: Dr. Jeff Black, The Wildfowl & Wetlands Trust, Slimbridge, Gloucester, GL2 7BT, United Kingdom; or Dr. Carol Terry, DLNR, Division of Forestry and Wildlife, 1151 Punchbowl St., Honolulu, HI 96813, USA.



Arabian oryx (*Oryx leucoryx*)
Photo: The Living Desert

The Arabian oryx originally ranged throughout most of the Near East, i.e., the Arabian Peninsula north to the Syrian Desert and Mesopotamia. Towards the middle of the 19th century, the oryx began to disappear from the northern parts of its range, aided by increased human activity in the desert and the availability of modern firearms. Automobiles and automatic weapons were the final straw, and wild populations succumbed to severe hunting pressure. Thus, in the 1960s the Fauna Preservation Society, in cooperation with the IUCN Species Survival Commission, initiated a captive breeding program with wild-caught individuals and a few zoo specimens. All oryx presently living in North America and European collections are descendants of these ten founder animals. Captive propagation has been quite successful.

Przewalski's horse (*Equs przewalskii*)
Photo: Zoological Society of San Diego

Przewalski's horse is the only true species of wild horse. Extinct in the wild, approximately 1,000 animals now exist in captivity. Although capable of interbreeding with domestic horses, the Przewalski's horse is a distinct species having 66 chromosomes (domestic horses have 64). A global management plan group exists that is coordinating the efforts on behalf of zoological parks to cooperate with nations within the historic range of the species to reintroduce it to its historic range in China, Mongolia, and Central Asia.

Kearney's blue star (*Amsonia kearneyana*)
Photo by Frank Reichenbacher

Individuals from private and governmental organizations collaborated to reintroduce Kearney's blue star (*Amsonia kearneyana*) to a site within its historical range in south central Arizona. Continuing threats to its survival include over-grazing, insect predation, and floods. When the U.S. Fish and Wildlife Service proposed Kearney's blue star for listing as endangered in 1988, only eight individuals in one population remained; today the reintroduced population numbers 136.

Captive Propagation in the Recovery of Whooping Cranes

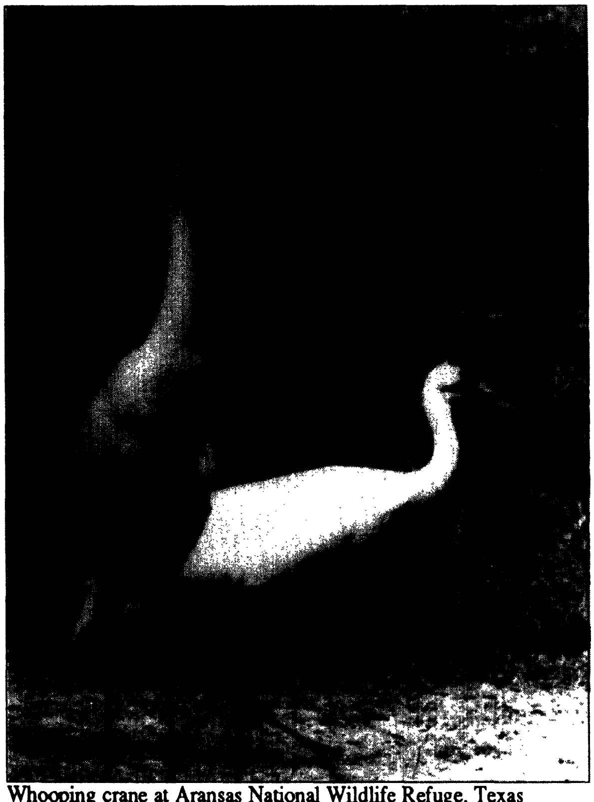
by
James C. Lewis

Reasons for Decline

The whooping crane (*Grus americana*) probably was most abundant during the Pleistocene. They were found from the Arctic Sea to the high plateau of central Mexico, and from Utah east to New Jersey and Florida. In the 19th century, the principal known breeding range was in parts of the prairie pothole region. A non-migratory population existed in southwestern Louisiana until the late 1940s.

Principal factors contributing to the species' decline in recent history are believed to be wetland destruction, hunting, and shooting for museum specimens. An estimated 500 to 700 birds survived in 1870, but they had disappeared from the principal United States breeding range by the 1890s. The last known nesting in southern Canada occurred in 1922. The only self-sustaining population now nests in Northwest Territories, at Wood Buffalo National Park. These birds winter in and near Aransas National Wildlife Refuge on the central Texas Gulf Coast where the population low was 16 birds in 1941.

Life history characteristics contributed to the decline and continue to hamper recovery of the species. Whooping cranes become sexually mature as early as age three, but on the average begin breeding at age five (Kuyt and Goossen 1987). The clutch size is two eggs, but a majority of the successful pairs arrive on the winter grounds with only a single young. Longevity in the wild is believed to be up to 25 years. Recovery of such a "K-selected" species is slow; by 1965 the wild population was only 65 birds. As a result, the species was federally listed as "endangered" in 1967.



Whooping crane at Aransas National Wildlife Refuge, Texas
Photo by W.F. Kubichek, USFWS

Captive Breeding Program

State and provincial agencies are important contributors to recovery actions within their areas of responsibility. However, the Canadian Wildlife Service (CWS) and the U.S. Fish and Wildlife Service (USFWS) are the primary agencies implementing the recovery action. The main concern of the two agencies is the single self-sustaining wild population which winters along the Gulf Intercoastal Waterway, feeding mainly on blue crabs and soft-shelled clams. The waterway is one of the busiest in the world. A barge accident could release petroleum or chemical products, destroy the food base, and kill many birds. As a consequence, a captive flock was needed to ensure the survival of the species and serve as a source of birds to start other wild populations.

Development of a captive flock began in the spring of 1966, when a single male was moved to Patuxent Wildlife Research Center (Patuxent). Earlier studies in Canada had indicated that a single egg might be removed from each two-egg clutch without significant harm to the wild population. Fifty eggs were moved from Wood Buffalo National Park to Patuxent (1967-1974), to start the population. Another 56 eggs were transferred (1982-1989) to improve genetic composition and to further build the captive flock.

Difficulties of captive propagation have made development of the Patuxent flock a long-term project. Starting with eggs meant waiting five to six years for sexual maturity, which was further delayed by conditions of captivity. Hatchability was best when eggs were incubated by sandhill cranes rather than in mechanical incubators. There were problems formulating adequate diets for the chicks, preventing parasite and disease infections, providing water and space, controlling light and temperature, preventing access by predators, and developing suitable animal husbandry techniques (Carpenter and Gabel 1984, Derrickson and Carpenter 1987). Many of the husbandry techniques developed for sandhill cranes were inadequate for the more fragile whooping cranes.

The problems mentioned have been solved to a large degree, but until they were solved, they contributed to losses of eggs, chicks, or adults to behavioral and physical abnormalities. The biggest single disease loss was the eastern equine encephalitis outbreak in 1984, when seven adult and subadult birds died, including several of the best breeders.

The first eggs were produced by a captive-reared female in 1975. Since then the number of females producing eggs has ranged annually from zero to five. All females have been artificially inseminated. In the 1980s, reproductive females on average produced 3.2 fertile eggs and fledged 1.4 chicks. The flock has produced 177 fertile eggs since 1975. Sixty-one fertile eggs were moved to Grays Lake National Wildlife Refuge for use in the attempt to start a wild population (1976-1984). To illustrate the difficulty in raising whooping cranes, the 166 fertile captive-produced eggs retained at Patuxent and the 71 Canadian eggs which hatched resulted in only 65 birds surviving in fall 1990. The average mortality rate among fledged birds has been eight percent annually.

Cross-Fostering and Reintroduction Efforts

Reintroducing an endangered species is a particularly challenging experimental process when the species must develop migratory patterns. Using an innovative research approach, Drs. Roderick Drewien and Elwood Bizeau of the University of Idaho suggested a cross-fostering experiment to restore whooping cranes to the northern Rockies. They suggested use of the abundant greater sandhill cranes (*Grus canadensis tabida*) nesting in Grays Lake National Wildlife Refuge in Idaho as foster parents for whooping cranes. Preliminary studies had shown that the typical two-egg clutch could be removed and another single egg substituted without the parent sandhill cranes abandoning the nest.

Project objectives included determining if: 1) sandhill pairs could successfully hatch, rear, and fledge whooping crane young from eggs; 2) foster-reared whoopers would accompany their parents to traditional wintering sites and return to their natal area; and 3) whooping cranes would survive and reproduce in these habitats. The project began in 1975, and egg transfers from Canada (216) and the captive flock (73) continued through 1988 (Drewien et al. 1989). Two hundred and ten eggs hatched, and 85 chicks survived to attain flight age. Mortality rates were excessively high in all age categories in the cross-fostered population in comparison to the natural population which nests in Canada (Garton et al. 1989). The population peaked at 33 birds in 1985. Thirteen of these birds still survive, including four females. The foster-reared whooping cranes learned the migration routes and stopover sites traditionally used by their foster parents (Drewien et al. 1989).

No cross-fostered whoopers, however, have paired and produced eggs. Since 1975, cross-fostered females of age four through ten years have passed through a spring nesting season on 26 occasions without breeding. Males and females appear to be associating together frequently enough, during winter and on spring migration stopover, to permit pairing at the times when pairing occurs in the natural flock. The absence of breeding may be due to improper sexual imprinting among the females.

"Reintroducing an endangered species is a particularly challenging experimental process when the species must develop migratory patterns."

Dr. Edward O. Garton, biometrician at the University of Idaho, working with Dr. Rod Drewien and others (Garton et al. 1989), modelled the cross-fostered population to predict when it might become self-sustaining. In the model they assumed: 1) the cross-fostered females would be breeding at the same rate as the females in Canada, and 2) survival of birds in their first year would be similar to that of first year birds in Canada. Despite these optimistic and unrealized assumptions, with the future transfer of 30 eggs per year, the population would only reach six breeding pairs after 50 years. "It is obvious from all scenarios modelled that egg transplants of less than 30 eggs per year will not suffice to establish a self-sustaining population in a reasonable period of time. Natural breeding will be essential to establish a self-sustaining population" (Garton et al. 1989). Fieldwork in the cross-fostering project is scheduled to end in summer 1991, when project personnel will concentrate on finishing their final contract report.

The self-sustaining wild whooping crane population now contains about 150 birds. As a consequence of the results in cross-fostering, the USFWS is now planning to use captive-reared whooping cranes to start a second self-sustaining wild population. The captive-reared birds would be conditioned for release in the wild. Special rearing methods include rearing the birds in isolation from humans and acquainting them with natural foods and conditions (Horwich 1989). This technique has been successful in supplementing the wild population of the endangered Mississippi sandhill crane (*Grus canadensis pulla*) which is nonmigratory.

The captive flock was split in 1989, and 22 birds moved to International Crane Foundation (ICF) in Baraboo, Wisconsin. This action was prompted after a series of health problems seriously threatened the entire flock at Patuxent. Flock division also provided an opportunity to utilize the crane avicultural expertise at ICF. The captive flock now totals 65 birds.

The Director of the USFWS and the Director General of CWS have approved fieldwork in Florida to determine the feasibility of establishing a non-migratory whooping crane population. Seventy-seven percent of the deaths or disappearances of the fledged cross-fostered birds occurred during spring or fall migration. A non-migratory flock would not be exposed to migration hazards. Another advantage is that a non-migratory population would be more concentrated year-round in a smaller geographic area. The more restricted movements would increase the opportunity for associations between birds that would be compatible mates, and increase the opportunity for egg production.

If whooping cranes are released in Florida, they will be reared in captivity and specially conditioned for release in the wild. Modelling studies suggest that a goal of releasing 20 cranes annually for at least ten years will increase the probability of successfully establishing a wild population. As Derrickson (1987) noted, "... the most significant historical factor limiting production has been the small size of the captive flock." To correct that problem, the USFWS has been endeavoring since 1988 to build the captive flock with captive-produced eggs and eggs from the Canadian population so captive pairs can support the release planned to

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Conservation of Crocodilians: The Release of Captive-Reared Specimens

by
F. Wayne King

Reasons for Decline

During the 1950s and 1960s, populations of crocodilians all over the world were depleted by indiscriminate hide hunting, killing as vermin, and habitat loss. During the late 1960s and 1970s, programs were launched in many countries to conserve these depleted and endangered populations (Bustard 1969, Cott and Pooley 1972). Some, like the American alligator, *Alligator mississippiensis*, were sufficiently abundant that when killing was prohibited, populations recovered. However, other species had become so endangered that simply ending the killing was not enough. The few adult specimens that remained were so scattered that breeding was almost nonexistent, and much of what little reproductive output did occur was lost.

Each generation a small number of young are lost to disease, but many more are lost to predators. Crocodilian nests are often raided by predators that feed on eggs: carnivorous lizards, mongoose, raccoons, bears, vultures, crabs, and ants. Hatchlings and young crocodilians are prey to herons, otters, fish, turtles, and mature crocodiles. From this variety of causes, between 50 to 90% of young crocodilians are lost during their first two to three years. In addition, as young crocodilians begin to mature, they often are killed by large males attempting to drive subadults out of their territories.

"Under normal circumstances probably less than 1 to 2% of wild [crocodilian] hatchlings survive to sexual maturity."

Under normal circumstances probably less than one to two percent of wild hatchlings survive to sexual maturity. Large crocodilians require eight to fifteen years to reach sexual maturity, so once a population is severely depressed, recovery is slow; years pass before the few young that survive predators and territorial males can in turn breed and contribute to the recovery.

The recovery process can be speeded up somewhat by incubating wild eggs artificially and rearing the young in captivity for later release back into the wild when they are large enough to resist most natural predators. Such "head starting" eliminates the large, early loss to predators and reduces the overall loss to the ten percent or less caused by disease. Territorial males will still attack subadult crocodiles as they approach mature sizes, but more individuals will reach that size class and presumably more will survive.

Reintroduction of the Gharial

The gharial, *Gavialis gangeticus*, was the subject of the first large-scale program of captive rearing and reintroduction for an endangered crocodilian species. Up until the 1950s and early 1960s, gharial were abundant in the large rivers of the Indian subcontinent: the Ganges, Brahmaputra, Indus, and Mahanadi rivers of India, Pakistan, Nepal, Bhutan, and Bangladesh, as well as the Kaladan and Irrawaddy rivers of Burma. Because of indiscriminate killing for their hides, as

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begin in Florida in the mid 1990s.

Aviculture techniques for whooping cranes are still being refined. Although our knowledge is incomplete, captive-reared birds are our only practical source of individuals to release in establishing a new population. I am confident that captive propagation and conditioning of birds for wild release can ultimately result in additional self-sustaining populations of non-migratory and migratory whooping cranes.

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vermin, and for use in folk medicine, as well as accidental drowning in fishing nets, and collection of their eggs for food, the gharial all but disappeared throughout most of its historical range. Preliminary surveys in the early 1970s indicated that as few as 200 adult gharials might survive in India, fewer then 20 in Bangladesh, and certainly no more than 20 in Pakistan. An estimated 65 to 70 adults survived in Nepal. The status in Burma remains unknown. So few adults remained that simply protecting them would not soon return populations to their former numbers.

Thus, in 1975, the government of India, with technical assistance from the wildlife division of the United Nations Food and Agriculture Organization and funding from the U.N. Development Programme, launched an all-out effort to save the gharial. Sanctuaries were established to protect the few scattered adults. For example, the Satkosia Gorge Sanctuary was declared on the Mahanadi River in 1976, at which time only five gharials could be found there. Eggs were collected from various remnant populations and hatched in government-operated hatcheries. By 1990, a total of 13 rearing stations for gharial were in operation in India (Choudhury 1990). Young were reared in captivity for up to three years before being released back into the wild.

By 1989, a total of 1,000 captive-reared gharials had been released in eight protected areas in India (Choudhury 1990). Half of that total, 550 captive-reared gharials, had been released into the Satkosia Gorge Sanctuary, which increased the resident population to 25 adults in 1988 (Kar 1989). Assuming that the original five gharials resident in the gorge were among the 25 recent residents, the survival rate for the released captive-reared gharials is four percent. More significantly, however, an adult female gharial began laying eggs in the Satkosia Gorge in 1984.

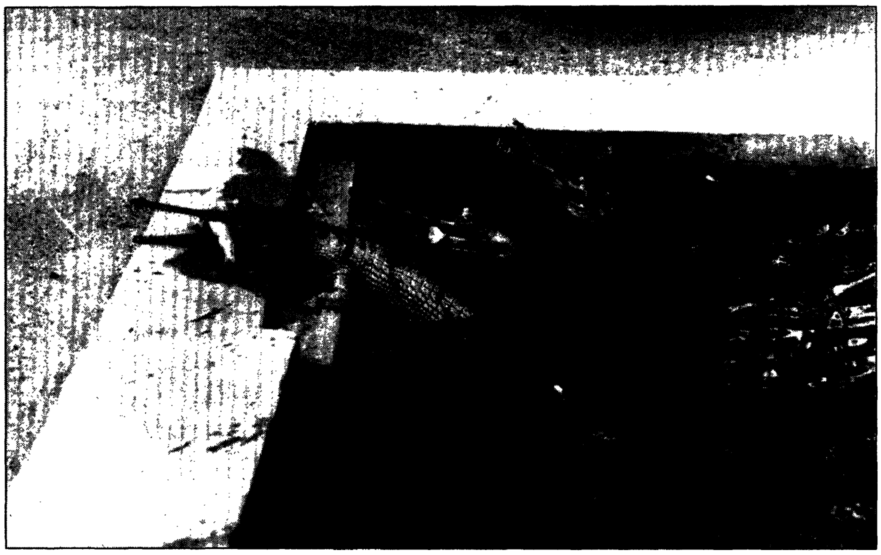
One of the problems hindering the reintroduction of wild populations by the release of captive-reared crocodilians is the movement of animals away from the release site. This is a problem for all crocodilians regardless of their habitats, but is especially critical for riverine species. Prior to their release, the pens in which they were reared constituted the home range of the captive-reared animals. With their sudden release into totally unfamiliar, albeit natural, surroundings, the young crocodilians set out to find their familiar home range. In marshes or lakes, they may simply wander away. In rivers, they may swim out into the current and be swept downstream. Some of the first gharials released in India were discovered later many kilometers downstream.

This search for home is not limited to young crocodilians. In the late 1960s, the New York Zoological Park shipped eight adult American alligators, *Alligator mississippiensis*, to Florida for release into the Everglades National Park. Included in the shipment were several animals measuring about three meters in length and weighing over 100 kg. Within two weeks of their release, all but two of the gators had moved away from the release sites. Even these two later disappeared.

The chance of success can be increased if the animals being released are maintained for several weeks in pens constructed in the natural wetland at the release site. This allows the animals to adapt to a new home range prior to venturing out into the natural habitat when the surrounding fence is later breached.

Other Crocodilian Reintroduction Programs in India

Since hatcheries and rearing facilities can be used for more than one species at a time, India did not limit its attention solely to the gharial. In 1975, India launched companion programs to conserve its other two species of crocodiles: the mugger crocodile, *Crocodylus palustris*, and the saltwater crocodile, *Crocodylus porosus*. Restocking newly established sanctuaries and suitable habitat with young hatched and reared in captivity played a major role in these programs as well (Choudhury 1990). By 1990, a total of more than 2,000 young muggers from 31 rearing stations were released in dozens of sites. The rearing stations still hold an additional 12,000 juveniles. The species now is breeding in the wild at 10 to 12 protected sites and in 25 rearing stations and zoos.


Young gharials (*Gavialis gangeticus*) at the Deori Rearing Centre, National Chambal Sanctuary, India
Photo by R.J. Rao

Similarly, nine rearing stations produced over 1,200 saltwater crocodiles which were released into the Bhitarkanika Sanctuary, where the species now breeds, and 24 have been released in other parts of the country. Relatively few saltwater crocodiles were found in Bhitarkanika's mangrove habitat when the sanctuary was declared, but a January 1988 census of the sanctuary counted 433 saltwater crocodiles, suggesting a 40% survival rate up to that time.

Crocodilian Reintroduction Programs Worldwide

Conserving crocodilians by captive rearing and restocking has not been confined to India. Similar "head starting" programs are a common feature of many crocodilian conservation programs:

American alligator, *Alligator mississippiensis*: In Louisiana, U.S.A., this program allows American alligator eggs to be collected from the wild under state license for supplementing production on alligator farms. It also requires farmers to return to the state for release back into the wild 30% of the hatchlings or 17% of the four-foot alligators produced from those eggs (Joanen 1989, Joanen and McNease 1990).

American crocodile, *Crocodylus acutus*: In Venezuela, a small number of captive-reared American crocodiles were released into the Jatira reservoir. Plans are also underway for releasing American crocodiles, produced by the captive breeding program on the Masaguaral ranch, into the Cuare National Wildlife Refuge (personal communication from Andres Seijas and John Thorbjarnarson).

In Colombia, INDERENA, the government agency responsible for wildlife conservation, requires the five farms licensed to breed American crocodiles to return five percent of their hatchlings to the government for restocking purposes.

Black Caiman, *Melanosuchus niger*: In August 1990, a total of 25 black caimans from a captive group on El Dorado ranch were released into Normandia lagoon adjacent to the Beni Biological Station where they could be protected (Thorbjarnarson 1990).

Broadsnouted caiman, *Caiman latirostris*: An experimental program is underway in Argentina for captive breeding of broadsnouted caiman for possible release.

Common caiman, *Caiman crocodilus spp.*: In Colombia, more than 30 farms have been established for captive production of caiman. INDERENA, the government agency responsible for wildlife conservation, requires the farms to return five percent of their hatchlings to the government for restocking purposes.

Gharial, *Gavialis gangeticus*: In 1978, Nepal established a "head starting" program in Royal Chitwan National Park, site of the largest surviving wild gharial population. Eggs were collected and hatched in a hatchery, and the young reared for later release at one to two meters in length (Maskey 1989). By 1990, approximately 1,000 gharials had been raised in captivity, and a total of 394 gharials released into the Nayarani, Rapti, Kali Gandaki, and Babai rivers (Maskey 1990). A 1990 survey of the Nayarani revealed 31 wild and 40 captive-reared gharials, representing 18% of the captive-reared gharials released there.

Pakistan is scheduled to receive gharial from the program in India so a captive breeding program can be initiated for restocking areas of protected habitat.

Morelet's crocodile, *Crocodylus moreletii*: Zoo Atlanta in Georgia, U.S.A., has been breeding Morelet's crocodiles for more than 15 years. Some of the captive-produced offspring have been returned to Mexico and Belize for release into the wild.

Mugger crocodile, *Crocodylus palustris*: Pakistan is planning a program for breeding mugger crocodiles in captivity for restocking former habitat.

Nile crocodile, *Crocodylus niloticus*: The government of Zimbabwe issues an annual quota for Nile crocodile eggs collected from the wild to supplement production on licensed crocodile farms. A percentage of the crocodiles hatched from the wild eggs are available to the government for restocking should they be needed. However, as long as wild populations are stable or increasing no return is required.

Orinoco crocodile, *Crocodylus intermedius*: Three centers in Venezuela are breeding and rearing the critically endangered Orinoco crocodile in captivity for release into protected habitat; the centers at the Masaguaral ranch and the El Frio Biological Station have been in existence for more than a decade, and the center at the Llanos University (UNELLEZ) for only about four years. In April 1990, 31 young Orinoco crocodiles were released into the newly declared Caño Guaritico wildlife refuge (Ayarzagüena 1990). Another release is planned to supplement the small population in the Capanaparo River in the Santos Luzardo National Park.

"Ranching gives local people a justification for tolerating the presence of large, aquatic carnivores that may threaten humans and livestock, and a motive for maintaining wetlands."

Philippine crocodile, *Crocodylus mindorensis*: Silliman University Marine Laboratory, Negros, is breeding Philippine crocodiles to restock protected habitat. In a collaborative effort with Silliman University, the species also is being bred in the Gladys Porter Zoo, Brownsville, Texas, U.S.A. Offspring produced at the zoo will be returned to Silliman University for release.

In addition to these programs already underway, restocking depleted habitat with captive-bred animals could make significant contributions to the conservation of several species of crocodilians:

Cuban crocodile, *Crocodylus rhombifer*: Two government-operated farms specialize in breeding Cuban crocodiles to preserve the species, to serve as a tourist attraction, and to provide hides for the market (Ramos Targarona 1989). The Cuban crocodile disappeared from the Lanier Swamp on Isla de la Juventud (Isla de Pinos) following the introduction of *Caiman crocodilus* (see Common Caiman below). If the feral caiman are removed, the swamp could be restocked with Cuban crocodiles from the farms.

Siamese crocodile, *Crocodylus siamensis*: This species is virtually extinct in Thailand and is extremely rare, if not extinct, in Indonesia. However, a captive population of approximately 8,000 pureblood Siamese crocodiles is held by the Samutprakan Crocodile Farm and Zoo, outside Bangkok (Luxmoore et al. 1985). If suitable habitat in Thailand can be identified and adequately protected, the farm could supply the necessary animals for reintroduction.

The release of crocodilians into the wild has not always been done for conservation. Some releases of exotic crocodilians into habitat outside their ranges have caused ecological problems for native crocodilians:

Common caiman, *Caiman crocodilus*: Although not part of a conservation program, the deliberate release or escape of pet caimans has resulted in the establishment of feral populations of common caiman in south Florida, U.S.A., Puerto Rico, and Cuba — see also Cuban crocodile above.

New Guinea crocodile, *Crocodylus novaeguineae*, and Philippine crocodile, *Crocodylus mindorensis*: Japanese occupation troops during World War II introduced New Guinea and Philippine crocodiles into Palau (Kimura 1968). The introduced species now compete with the small native population of saltwater crocodile, *Crocodylus porosus*.

Crocodilian Ranching

With few exceptions, "head starting" programs involving crocodilians are an adjunct to efforts to conserve wild populations through ranching. Ranching involves protecting adult breeders in the wild and harvesting their eggs or hatchlings to rear in captivity for later commercial slaughter. Money earned from the sale of hides and meat benefits the crocodile rancher, and ranching in turn benefits crocodilian conservation.

As ranching depends on wild eggs or hatchlings for its stock, it can only be sustained as long as wild adult breeders are sufficiently abundant to support the harvest. This forces ranchers to be concerned about the wild populations. In contrast, farming, which involves the hatching of eggs laid in captivity from matings of captive parents, is not dependent on the wild population and does not directly benefit the wild resource.

Ranching gives local people a justification for tolerating the presence of large, aquatic carnivores that may threaten humans and livestock, and a motive for maintaining wetland habitat. For example, villagers along the Sepik River of Papua New Guinea (PNG) eat crocodile eggs. However, under the PNG conservation program, crocodile ranchers pay landowners the equivalent of U.S. $1.50 for each crocodile egg collected on their land. In addition to the thousands of dollars they earn from selling eggs, the landowners are given one hen egg for each crocodile egg collected. The money pays the landowners for protecting crocodiles in the wild, and the hen eggs replace crocodile eggs on the dinner table. Annual censusing indicates that the wild population is increasing, and that collecting crocodile eggs solely for human consumption has all but disappeared.

Similarly, in the U.S.A., conservation programs in Florida, Louisiana, and Texas allow landowners to earn money from the sale of alligator eggs or hatchlings collected from their wetlands. Where zoning and environmental laws allow them to do so, landowners could earn more money by turning their wetlands into agricultural lands or into housing developments. However, many landowners enjoy the out-of-doors, and the money they collect from allowing the harvest of alligator eggs or hatchlings is sufficient excuse to maintain the wetlands.

This money that ranchers and landowners in PNG, the USA, and elsewhere earn makes them less tolerant of poachers who would overexploit the resource or of developers who would destroy the habitat. Crocodilians are rather like the mythical "goose that laid the golden egg." In the past, crocodilians, like the goose, were destroyed by poachers and hunters who overexploited the resource. Ranching is changing that. Now in order to exploit the resource on a sustainable-yield basis, ranchers and landowners must protect wild crocodilians and their habitat.

Obviously, headstarting and captive breeding of crocodilians for conservation purposes differ greatly in their objectives from ranching for commercial profit. However, some of the same techniques apply, and all can be used to benefit wild populations. Thus, in the case of crocodilian conservation, captive breeding combined with economic incentives has been quite effective in actually increasing wild populations, and perhaps more importantly, in providing motivation to governments and local peoples to protect wild individuals and habitat.

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Captive Breeding and Reintroduction: Recovery Efforts for the Virgin Islands Boa, *Epicrates monensis granti*

by
Peter J. Tolson

History

The Virgin Islands boa, *Epicrates monensis granti*, was first discovered by Major Chapman Grant on the island of Tortola, British Virgin Islands (Grant 1932). Erroneously described by Stull (1933) as a subspecies of the Puerto Rico boa, the snake was designated as endangered under the U.S. Endangered Species Act listing for E. *inornatus* in 1973. Taxonomic work by Sheplan and Schwartz (1974) later demonstrated that the snake was actually a subspecies of the Mona Island boa, *E. m. monensis*. In 1980, the legislative status of the snake was finally clarified by the U.S. Fish and Wildlife Service (USFWS), and it received endangered species protection under the correct nomenclature (USFWS 1980).

Apparently never very common, its extremely disjunct distribution as a Puerto Rico Bank endemic (Nellis et al. 1983, Mayer and Lazell 1988) provides evidence for a long history of extirpation and decline on numerous islands and cays (low islets of coral or sand) of the Bank since the Pleistocene. It is believed that a variety of factors have contributed to the present dire status of this snake. Habitat destruction was among the earliest of perils, as widespread areas of the Virgin Islands were denuded and periodically burned for the cultivation of sugar cane. The introduction of exotic predators such as the roof rat, *Rattus rattus*, the house cat, *Felis catus*, and the Indian mongoose, *Herpestes auropunctatus*, also had serious adverse effects on boa populations. Climatic changes during the Pleistocene (Pregill 1982), eustatic events and the subsequent genetic isolation of several populations, and the stochastic processes associated with the fragmentation of a once contiguous range have also doubtless contributed to the decline of this taxon (USFWS 1986, Tolson 1988).

Captive Breeding and Reintroduction

In 1985, in cooperation with the USFWS, the Departmento de Recursos Naturales de Puerto Rico (DRN), and the Division of Fish and Wildlife, U.S. Virgin Islands (USVIFW), the Toledo Zoological Society (TZS) initiated a captive breeding program for this species. The focus of the TZS program is the production of snakes for reintroduction efforts. The basis for a management strategy was developed under the auspices of the USFWS Recovery Plan (USFWS 1986), and included two years of field research on the ecology and distribution of the snake in Puerto Rico and the Virgin Islands. Data continues to accumulate from wild populations through an ongoing mark-and-recapture study of over 200 marked individuals on La Cordillera. These efforts were coupled with a comprehensive reproductive research program using congeners as models which established the proximate environmental and social factors critical to reproduction in this species (Tolson and Tuebner 1987, Tolson 1989).

Virgin Islands boa juveniles born at the Toledo Zoological Gardens   Photo by Art Weber

The reproductive program has thus far been successful: 48 offspring have been produced by ten founders at the TZS, of which 40 have survived. Representatives of two distinct demes — La Cordillera, Puerto Rico, and St. Thomas, U.S. Virgin Islands — are now maintained in captivity. As each deme exhibits a distinct coloration, it was decided to maintain deme integrity in the captive snakes and manage them as two separate populations, i.e., individuals captured on La Cordillera are not crossbred with stock originating from St. Thomas. Unfortunately, the two male and three female boas collected from St. Thomas have not yet reproduced; two of these females are sexually immature and the third expelled infertile ova after her first reproductive attempt in captivity. The rarity of snakes on St. Thomas makes them very difficult to obtain, and the actual size of the population there is unknown.

To aid in captive management, the TZS maintains a regional studbook for the species under the auspices of the American Association of Zoological Parks and Aquariums (AAZPA). In addition, a petition to establish a Species Survival Plan for this snake has been submitted to the AAZPA. (Species Survival Plans are cooperative management programs in which several zoos pool resources to develop captive breeding strategies for endangered wildlife.)

Critical Factors Affecting Reintroduction Success

Reintroduction is perceived to be a reasonable strategy in the recovery of this species because of the availability of protected, relatively undisturbed cays which exhibit a number of community attributes identified by principal components analysis as typical of habitat where this species occurs (Tolson 1988). However, any reintroductions to the wild will involve only those individuals originating from a source population which is closest to the release site. A lack of correlation between availability of captive-bred individuals from specific localities and site preparation has been the greatest barrier to immediate reintroduction attempts. Captive-born offspring from St. Thomas founders are not yet available for release even though there are two rat-free cays in the Virgin Islands that could serve as reintroduction sites.

In fact, in the opinion of the author, the greatest single problem to overcome in any release program for this species is the eradication of exotic mammals from potential reintroduction sites. Rat and cat control programs have been initiated at two potential reintroduction sites on offshore cays, one near Puerto Rico, and another in the Virgin Islands. Snakes cannot be released at these localities until control programs have been proven to be successful. Recolonization, deliberate or accidental, of *Rattus* or *Felis* to reintroduction sites could doom any reintroduced populations. A third suitable cay, Steven Cay, already cleared of rats by the USVIFW in 1983, is also suitable for reintroduction. The site identified by the author (Tolson 1988) on Cayo Icacos as a primary candidate for reintroduction was severely damaged by Hurricane Hugo in September 1989, and an alternate site on La Cordillera had to be located.

Evaluation of Reintroduction Strategies

There are other problems which must be overcome if reintroduction is to succeed in this species. Genetic management of the population may also be of some concern, as completely random matings within the captive population may be difficult to achieve due to the reluctance or inability of some potential founder snakes to reproduce. Two males captured in 1986 from the wild on La Cordillera have yet to produce offspring in captivity, despite numerous reproductive opportunities. No genetic analysis exists for any of the wild populations; the degree of heterozygosity within populations and the degree of relatedness between isolated populations is completely unknown. Until these data are generated, genetic management will be haphazard.

But there are several attributes of this program which should ultimately result in its eventual success. There is a strong base of research on which to base management decisions (see Tolson 1988, Tolson and Tuebner 1987, Chandler and Tolson 1990). The environmental and social parameters necessary to induce reproduction can be duplicated fairly easily in captivity and the husbandry of the species is not difficult. Space requirements are minimal, and the survivorship of captive neonates is extremely high — exceeding 83% at the TZS. Although *E. monensis* reproduces biennially, longevity may exceed 20 years, lifetime female reproductive output is potentially very high, and generation times can be extended to preserve genetic diversity in the captive population. In addition, demographic research has demonstrated that this species can exist in high densities on small islands. With a probable high degree of inbreeding in several wild populations, the genetic load of deletrious alleles is apparently not high.

Several additional activities are underway to prepare for reintroduction attempts. Animals slated for release are fed live adult male crested anoles, *Anolis cristatellus*, to insure the snakes are able to capture and feed upon their natural prey. Veterinary protocols for screening of animals to be released are being developed at the TZS. The initial releases will probably involve radiotracking of individuals implanted with transmitters to assess to what degree captive-raised individuals duplicate the behaviors of wild snakes.

Federal, Commonwealth, Territorial, and Society management personnel have a five-year history of successful cooperation in recovery efforts for this species. With continued commitment from these agencies, it is believed that future management actions will be successful as well.

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Release and Translocation Strategies for the Puerto Rican Crested Toad, *Peltophryne lemur*

by
Robert R. Johnson

Introduction

Amphibians, with both aquatic and terrestrial stages in their life cycles, are particularly sensitive to environmental stresses, and to contaminants acquired in their diet (herbivorous in the larval stage and insectivorous as an adult) or through their permeable skin. As a result, there has been a recent concern over what is perceived to be a global decline in amphibian populations.

Release and translocation programs for amphibians, which have reproductive strategies unlike most other vertebrate species, are not well developed as of yet. An examination of the progress of the Puerto Rican crested toad program may be illuminating for those involved in or contemplating similar amphibian release programs.

Current Status of Toad Populations

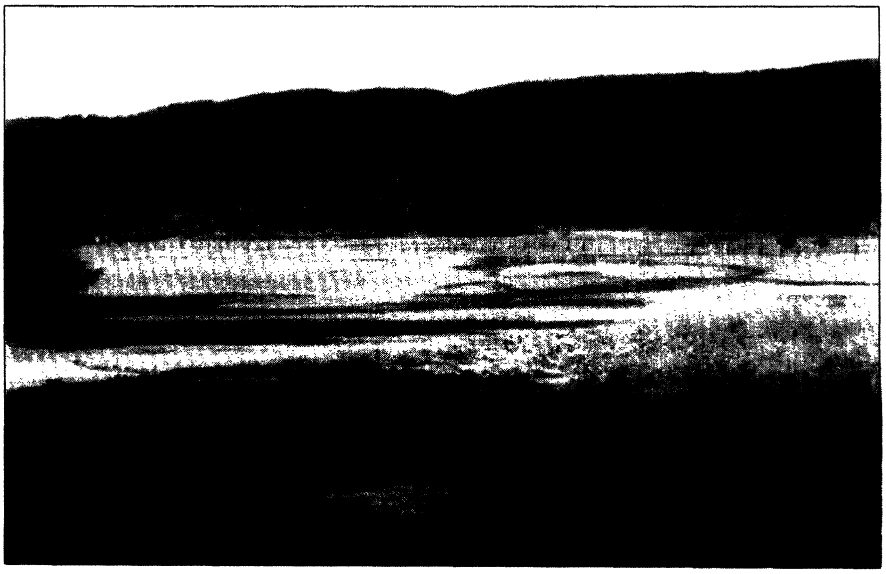
Perhaps the most significant factor underlying the captive breeding and release of Puerto Rican crested toads is the precarious state of wild populations. Although the rarity of this species has been questioned since its rediscovery in 1967, no one has demonstrated that there exists any more than 25 individuals in the northeast coastal plane near Quebradillas, and 3,000 individuals in Guanica Forest in the southeast. Although Moreno (1985) has described the Guanica population as healthy, numerous and stable, an examination of utilized breeding sites illustrates the present precarious status of this species.

Northern Population

In the north, *Peltophryne lemur* is known to have bred in four concrete, walk-in cattle troughs located near Quebradillas. Each of the four sites flanks the same dry gully which carries storm runoff a short distance before terminating at the sea. This ravine is utilized by toads as a refuge and as a migration route to the cattle troughs. No more than 25 individuals have been seen at any one breeding event. If this represents the number of breeding adults, this population is not large enough to sustain itself, given the susceptibility of small populations to stochastic events and population demographics. In fact, skewed sex ratios, reported by Rivero et al. (1980), may hasten the extirpation process.

None of the northern troughs are protected, and cattle still utilize them as water sources. Each is highly eutrophic, and spraying of nearby pastureland for cattle ticks may result in high levels of pesticides in field runoff, which is the only source of water for the troughs. One of the nearby pastures has even recently been utilized as a junkyard for the storage of automobiles. The recent acquisition of land for hotel development near Isabella has increased land value in this area — even small land holdings are for sale at $6,000 per acre.

Individual sightings of toads in this area of development should result in protection or enhancement of the breeding habitat as part of the development proposal. Unfortunately, development may destroy breeding areas or essential toad habitat before their significance is identified. The lack of field surveys and reliable field reports to confirm the presence of toads or toad habitat hinders effective decision-making in this regard.



View of breeding pond in Guanica Forest. Toads live in the hills behind the beach. Posts were erected by the Forest Manager to protect the breeding pond. Photo by B. Johnson.

Southern Population

In 1984, Miguel Canals, the Guanica Forest Manager, discovered a large population of crested toads in a flooded parking area of Guanica Forest on the southern coast. This population consists of approximately 3,000 individuals which reproduce in one pond located 100 meters from the sea. Despite its protected location in the Tamarindo sector of the Guanica forest, the pond is under threat of inundation by sea water during tropical storms. Indeed, a direct hit by a hurricane could result in permanent loss of the breeding pond if the small section of barrier beach is cut into by heavy seas. Furthermore, the beachside parking lot receives heavy day use throughout the year. As the water in the parking area is continuous with that in the breeding area, any contamination of soil or water might affect developing toads.

When 13 inches of rain fell in a 24-hour period during Hurricane Gilbert in 1985, several ponds were breached as water rushed down from the surrounding hills. One such pond, which sustained a limited number of breeding pairs, was washed out. Miguel Canals witnessed many pairs of toads washed out to sea and has not observed any toads returning to this pond during the subsequent breeding events. In June 1990, while radiotracking crested toads which had bred for the first time in almost two years, we visited this pond and observed *Leptodactylus albiaris* (white-lipped frog), but no *Peltophryne lemur*.

The natural beauty of the beaches in properties adjacent to the Guanica Forest has increased pressures for development. Concerns have been expressed over the impact of support infrastructures and the need to improve accessibility, which will be necessary if hotel development occurs. Additionally, as with the northern population, the known populations and breeding habitat of this species in the south cannot be considered secure.

History of Releases

In 1966, six northern toads were found — the first sightings since 1931. After a 1982 meeting at the Jardin Zoologico de Puerto Rico to discuss the support role of captive propagation, several toads were found during a visit to one of the cattle troughs. These individuals later reproduced and their offspring were sent to the Buffalo and Indianapolis Zoos. The crested toad subsequently became the first amphibian to be included in the American Association of Zoological Parks and Aquariums (AAZPA) Species Survival Plan (SSP).

After a hormonally induced breeding at the Buffalo Zoo in 1983, offspring were released into the cattle trough in which their parents were discovered (Miller 1985). Northern toads bred at the Buffalo and Toronto Zoos in 1984 and 1985 were released into a small human-made pond in Cambalache Forest Reserve. Cambalache forest was chosen after consultation with Puerto Rico Department of Natural Resources (DNR) staff ecologists, who requested the release of as many toadlets as possible from captive-bred offspring still the property of the DNR as part of breeding loan agreements. This northern site was within the historic range, did not have an existing toad population, and was characterized by dry forest and exposed limestone, considered to be an essential component of the type habitat.

Male Puerto Rican crested toad — Photo by B. Johnson

In 1985, 20 toadlets were collected from leaf litter surrounding the Guanica breeding pond by representatives from Metro Toronto Zoo and DNR. These toadlets formed the basis of a southern founder population to be managed under the AAZPA SSP program. Breedings of these founders resulted in the recommendation by the DNR for two additional releases of toadlets into an area of Guanica forest selected for the presence of water which collected in borrow pits and because it was considered to be outside the migratory range of toads using the beach-side breeding pond. The only evidence of survivorship has come from a 1989 discovery of two animals released as part of a group of 640 in 1988.

Radio-tracking of 12 captive-raised toads in 1989, and 12 wild, post-reproductive toads in 1990, has provided data on survivorship, home range, and habitat use. The small size of released toadlets makes followup on the success of introductions or releases difficult, but nonetheless important. To date, the only measure of success has been the presence or absence of toadlets at release sites.

Conservation In Action

The importance of the discovery of the Guanica population by Miguel Canals cannot be underestimated, nor can the series of actions he initiated to secure this site for subsequent breedings: the ditch which each year drained the flooded parking area in which the toads bred was dammed; a series of posts were planted to prevent cars from driving into the breeding area and destroying vegetation used by toads during egg laying; and during the three-week breeding period the road to the parking lot is closed to prevent water contamination. A public information campaign through local schools and newspapers has resulted in local acceptance of the changes which have occurred in this high-use area to ensure the survival of the toad. Thus, the survival of the species to date has been largely dependent on the efforts of one individual.

Current Status of Release Program

The long-term survival of the crested toad is dependent upon protection of existing breeding sites and establishment of satellite populations. However, there are no plans for further releases into the wild until suitable sites with release ponds, either natural or human-made, can be identified.

In 1990, a Metro Toronto Zoo research team collected data on pH, salinity, calcium and temperature at Guanica to characterize the chemistry and ecological characteristics of the breeding ponds. This pond profile data is essential to allow managers to maintain the character of existing breeding ponds in case of any sudden or long-term adverse changes, and to identify any other existing ponds suitable for crested toad reproduction. If no appropriate ponds are located, it will be necessary to construct breeding ponds which include the essential components of the present pond. It is anticipated that new ponds, whether naturally formed during rains or constructed to fill during rain, would receive tadpoles from two sources. Some tadpoles would be translocated from the present breeding pond to adjacent ponds outside the migratory range of the toad, and a second pond would receive captive-bred tadpoles returned to Puerto Rico for release.

These satellite ponds would serve as reservoirs of toads in the event of stochastic destruction of the present breeding pond. As well, the ponds to receive tadpoles would allow the comparison of the success of translocations of tadpoles from the Guanica pond with releases of captive-bred animals. This data is important for future releases of this and other imperiled species in order to make the most effective use of small populations in future conservation programs.

Puerto Rican crested toad wearing harness which holds radio transmitter
Photo by B. Johnson

"The success of reintroduction programs must be considered in a time frame of perhaps ten or more years, and after a number of releases."

Strategies for Future Toad Releases

As a result of our experience to date, we suggest the following points be considered in future release programs:

1) Given the need to locate a natal pond for a breeding generation of toads, success in the short term should not be anticipated. The success of introduction programs must be considered in a time frame of perhaps ten or more years and after a number of releases.

2) The chemistry and ecology of existing ponds must be characterized to test the suitability of future breeding sites (satellite ponds), and to monitor long-term changes which may negatively impact existing breeding ponds.

3) The establishment of new breeding populations might be improved if the translocation of wild tadpoles and release of captive-bred toads was considered in tandem. The success of each approach would provide comparative data for future release or translocation.

4) Several founder populations are required for the repeated release of numbers of unrelated lines. The status of northern populations is of particular concern given the limited number of toads in captivity and known to exist in the wild.

5) It would appear that the number of breeding animals presently known from the north has fallen below the minimum viable population size necessary to sustain a population. Further study will indicate the need to augment populations centered at each of the four breeding ponds as an appropriate management option, although the extirpation of northern populations in the Quebradillas area seems imminent. The establishment of satellite populations may be an alternative to augmenting existing populations from captive-bred stock.

6) If northern lines are extirpated, southern animals may be translocated to the historic range. Studies of mitochondrial DNA (Anna Goebel/Geoff Minton, University of Colorado at Boulder) are underway to determine the extent to which northern and southern populations have diverged.

7) Genetic swamping can be anticipated with the release of up to 5,000 headstarted offspring from a single captive breeding event. For this reason, it is necessary to limit representation from each founder and to utilize as many founders as possible. This is especially important if the wild population has suffered a severe reduction in numbers or there is evidence of bottlenecking. In areas where the existing population has been extirpated, the release of such genetically related lines may be the only alternative, even if long-term environmental change or stress has been a factor in the extirpation of the species. Genetic swamping may be reduced if releases involve tadpoles, the lifestage during which mortality is highest.

8) Given the high mortality rate in wild amphibian populations, the number of animals released should be based on the projected survival rate and the carrying capacity of the release environment.

9) It is preferable to keep all animals intended for release in conditions which simulate the environmental stresses of the release environment. Short-term holding in large outdoor enclosures (preferably at the release site itself) would facilitate the initial orientation process and physiological adjustment to arid environments. The importance of map and compass orientation from and to natal ponds requires further testing. The controlled release of offspring to test the effectiveness of introduction procedures and stock may be advisable before implementation over larger areas.

10) It may be necessary to monitor and, if present in high densities, eliminate mongoose, rat and marine toad popula-

tions in breeding or release areas. Released toadlets may have to be protected for short periods of time from high densities of lizard and avian predators, at least until they have found rocky refuges.

11) Captive breeding and translocations must not be ends in themselves. Protection and restoration of existing habitat are essential components of conservation programs involving releases of captive-bred animals. The natural colonization of human-made or enhanced ponds, strategically placed in patches extending away from the prime breeding sites, may be a valuable adjunct to the existing population, rather than the removal and return of animals. These satellite ponds may be colonized much more rapidly than those seeded with naive animals returned from captivity, and their value may be proportional to their proximity to protected natal ponds. This is an exciting aspect of species conservation as it involves the new science of restoration ecology, and would require the enhancement or creation of habitat on an experimental basis.

12) The demographic profile for introductions will be weighted towards numerous tadpoles or newly metamorphosed toadlets since, initially, these are the fodder upon which selective processes occur. Some consideration should be given to retaining a limited number of individuals in captivity until they are of reproductive age before release. This would eliminate the life stages in which highest mortality occurs before the actual release, but perhaps reduce the fitness of the animals since the adults are naive to the home environment (Johnson and Paine 1989). Which of these considerations are most relevant to the production of reproducing animals has not yet been established. To release both lifestages would in effect be a hedging for survival until reproduction can occur.

13) The release of captive-bred animals along with protection and, if necessary, enhancement of existing breeding habitat can create publicity for the fate of this and other threatened species. Public education must be integrated into the conservation program. Other species may benefit from increased awareness of the reasons for the decline of popular or target species. For example, Metro Toronto Zoo is producing a poster, to be circulated by DNR, which will relate the toad to its unique habitat in Guanica forest. It invites Puerto Ricans to identify other populations of the toad, and appeals for protection of essential habitat and other threatened species.

"The exemplary successes of this program may in fact create an environment in which the preservation of lesser-known species can be anticipated."

14) The AAZPA Puerto Rican Crested Toad SSP Program and conservation within protected reserves in Puerto Rico must be integrated. We may never fully understand the diversity of relationships between toad and environment which will ensure the continued survival of this island endemic. In light of our limited understanding of the processes of extinction of free-ranging populations, we must continue to manage populations and habitat which may have been modified either intentionally or unintentionally by human impact.

15) With the establishment of additional breeding sites, this could become a short-term conservation project. The winding down of the conservation effort necessary at this time to ensure the survival of the crested toad would allow the allocation of limited captive breeding resources for other species in need of intensive management. The exemplary successes of this program may in fact create an environment in which the preservation of lesser-known species can be anticipated. The involvement of zoo-bred animals released to a protected environment ensures that captive space will be available for the continued survival of amphibian biodiversity globally and in Puerto Rico.

Summary

The success rate for non-game translocations will be low until trials can be conducted and analyzed. A period of relatively poor establishment must be anticipated which should not negatively impact the consideration of species for further introduction attempts. The release of captive-bred animals and habitat restoration are management tools which will increasingly be the last resort for a number of species. The potential of success in release programs must be considered as only one management strategy among others in stemming the rapid and catastrophic loss of amphibian species diversity.

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Lepidochelys kempii hatchlings Photo by M. Graham Blake

Mexico has been studying the decline of *Lepidochelys kempii* at Rancho Nuevo, Tamaulipas, since the mid-1960s. In 1978, Mexico, the U.S. Fish and Wildlife Service, and other U.S. agencies began a binational effort to save the species. For the last ten years, the U.S. Field Assistance effort has been administered by personnel of the Gladys Porter Zoo. In 1990, 967 nests were protected with 66,663 hatchlings released at Rancho Nuevo and an additional 2,025 shipped to the National Marine Fisheries laboratories at Galveston for headstarting. It is hoped that protecting the hatchlings from land living or aerial predators while crossing the beach will enhance the species chances for future recruitment.

Central Florida Scrub: Trying to Save the Pieces

by

Susan R. Wallace

Introduction onto protected land will be the only way to save many of Florida's rare endemic plants. As the scientific and philosophical discussions continue, and while the technology is still being developed, time is running short.

The Ancient Scrub Community

Because of its unique geological history, Florida is home to an unusually rich assortment of endemic plants and animals. An area of high ground down the center of the state, the Lake Wales Ridge, was a series of islands during the Pleistocene interglacials. As sea levels rose and fell, and Florida was shaped and re-shaped, the isolated populations on these islands evolved to produce one of the highest rates of endemism in the continental United States.

On the crest of the Ridge are remnants of the longleaf pine/wiregrass community which was once common in the Southeast. Here, however, it is punctuated with rarities like the delicate annuals: wide-leaf warea (*Warea amplexifolia*), Carter's mustard, (*W. carteri*), and the dwarf plum (*Prunus geniculata*).

At a slightly lower elevation, presumably the site of ancient shorelines, are patches of scrub habitat which represent the state's richest flora and most endangered community. These areas are characterized by deep, dry sands of low fertility, intense hot sun, a short summer rainy season, and long droughts in winter. Each scrub patch is slightly different, both biologically and aesthetically. Generally there is a cover of dwarfed sclerophyllous oaks, palmettos (*Serenoa repens*, *Sabel etonia*), and cactus (*Opuntia* spp.). Often, but not always, there is an overstory of sand pine (*Pinus clausa*). The soil may be yellow sand or the dazzling, snow-white sand found on many of Florida's present-day beaches.

Christman (1988) calls these areas "ancient scrubs" to distinguish them from similar coastal scrub communities which lack the same concentration of endemics. Some ancient scrub sites are so rich that many rare species, both plant and animal, occur together. On the one undisturbed site for the rare Florida ziziphus, *Ziziphus celata* (the other two sites are in "improved" pasture), there are 23 endemic plant species (Delaney et al. 1989), of which 11 are considered at risk.

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However, most of these rare endemics have extremely narrow ranges, even within the scrub communities of the Ridge (Christman and Judd 1990). There is no one site, nor even a group of sites, which contains the entire range of heterogeneity along the Ridge. [Coastal scrub communities, particularly along the Atlantic and northern Gulf coasts, also contain unique scrub species, although not as many. And again, the ranges are generally very narrow.]

Species Decline

Habitat destruction has been widespread in Florida, and these areas of high ground—the endemics' home ground—have been particularly hard hit. Two-thirds of the ancient scrubs in central Florida have been destroyed (Christman 1988). Much of the Lake Wales Ridge has been cleared for citrus; the deep sands and (once) mild climate are ideal for citrus and large portions of the area have been planted in mile after mile of groves. Ironically, much of this citrus -- at least the northern half -- is now destroyed, wiped out by three major freezes in one decade. Along the Ridge, where rare species may once have been common, are now acres and acres of dead groves and ruderal vegetation.

Cattle ranching also has been responsible for clearing large portions of the state. Additionally, human development, including homes, shopping malls, tourist attractions, and everything that goes with them, continues at a rapid pace. Major highways travel along the crests of upland areas — Highway 27 atop the Lake Wales Ridge and Highway 1 along the Atlantic Coast Ridge — right through some of the rarest plant habitat in the state.

Florida's initial efforts at land conservation and acquisition had been directed towards wetland preservation. Now attention has turned to the remarkable species diversity of the high ground areas. A state land-purchasing program and The Nature Conservancy have recently set aside several large tracts of pristine scrub habitat, and more sites are under consideration.

But all of this may come too late for many rare species which exist now in only a few remnant patches of scrub—in a vacant lot, along a fence row, in a ditch, behind a gas station. Many endangered species live only in small populations on land that is too urban to protect and too expensive to buy. These plants have no legal protection, no management, and nowhere to go, surrounded by development and vulnerable everyday to bulldozers, cement trucks and herbicide spray.

For instance, Lakela's mint (*Dicerandra immaculata*) is known only from a ten-mile stretch of Highway 1, less than one mile wide, right in the middle of an expanding urban area. Although some seemingly robust populations still exist, they are on land with expensive highway frontage and "For Sale" signs.

The related Garrett's mint (*Dicerandra christmanii*) was known from five sites, one of which was recently destroyed. Other populations are quite small; the largest is on part of a defunct subdivision. Longspurred mint (*Dicerandra cornutissima*) grows in fence rows of an interstate highway and along road-cuts in a nearby housing development.

Wedge-leafed eryngium (*Eryngium cuneifolim*) is known from 20 sites along the Ridge, ranging a distance of less than 20 miles (Christman and Judd 1990). One protected population grows at Archbold Biological Station. The other plants grow on private land and roadsides; a few forlorn remnants hang on in the sand behind a shopping center.

Ziziphus celata is known from three sites, a total of five clones. Only one site is undisturbed. All are privately owned.

This sad litany mentions only a few examples. Some species now have a few populations on protected sites — which seems reassuring when compared to the species which have no protection at all. But realistically we are still looking at numbers too few and areas too small to consider any of the species secure.

Propagation and Reintroduction Efforts

Bok Tower Gardens' Endangered Plant Program was begun in response to this critical situation. Four acres have been set aside, away from the public area of the Gardens, to house the ex situ collection. A new greenhouse and a grid of special planting beds have been constructed where genetically representative collections of the plants can be propagated, grown, and monitored regularly.

Now, with four years of experience and 30 species accessioned into the collection (to one degree or another), the limitations of ex situ preservation have become clearer and the need for introduction projects more imperative.

Some species simply won't respond to cultivation at all. Scrub lupine (*Lupinus aridorum*) is such an example. Few cuttings will root. Those that do root die immediately after transplanting, unable to tolerate any root disturbance. Seeds will germinate, then dwindle and die within a few weeks, even under numerous permutations of cultural regimes. Yet natural seedling generation, where conditions are just right, appears adequate, though never robust. Clearly we don't understand what this plant needs, and its requirements are very exacting. With incomplete knowledge, little time, limited resources, and very few seeds to work with, our intuition is to put the few seeds available on carefully chosen sites and let nature take over. Fortunately, not all the species we work with are this humbling.

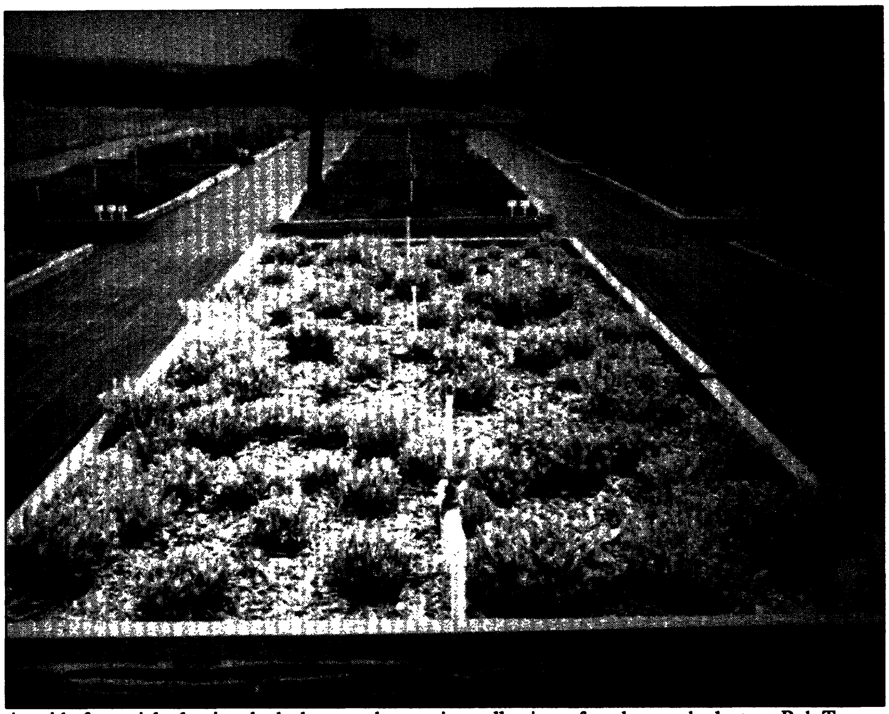
This central Florida roadside in the white sand scrub of a defunct housing subdivision is home to four federally listed plant species. Photo by S. Wallace

"A state land-purchasing program and The Nature Conservancy have recently set aside several large tracts of pristine scrub habitat. . . But all of this may come too late for many rare species which exist now in only a few remnant patches of scrub -- in a vacant lot, along a fence row, in a ditch, behind a gas station."

Other species can be maintained in cultivation, but are so short-lived, with an attrition rate so high, that they require an overwhelming commitment of man hours, greenhouse bench space, and land area to continually repropagate. This is especially true when each clone has to be tracked individually in an effort to maintain a genetically representative population.

The four *Dicerandra* species in our collection, whose natural strategies rely more on volume than longevity, present just such a problem (Wallace 1990). Seedling generation for this genus has been disappointing under greenhouse conditions (for reasons not yet clear), although it is frequently robust in the wild. We have established an experimental population of cutting-grown Scrub mint (*Dicerandra frutescens*) in the protected buffer area surrounding the Gardens. These were taken from a small population very near the Gardens, found in a vacant lot of a subdivision, and planted on a site which seems to approximate the original. (The associates, however, are not exactly the same, thus one thing we expect to find out is how much this matters.) If this population prospers and seeds itself, we will look for protected sites within the ranges of the other species.

A naturally occurring population of *Warea amplexifolia* grows within the wiregrass of Bok Tower Gardens' Pine Ridge Preserve area. (Of the three other



A grid of special planting beds houses the ex situ collection of endangered plants at Bok Tower Gardens. In the foreground is *Chrysopsis floridana*. Photo by S. Wallace

populations of this species, one is on state park land, another in a vacant lot, and the fourth in the path of a land-eating sand mine.) This fall-flowering annual crucifer, with dainty lavender blooms, shows a marked variation in population size from one year to the next, apparently forming a fluctuating seed bank in the soil. (The related Carter's mustard, *W. carteri*, has appeared on sites where it had not been observed at all for several years previously (Bissett 1988).) There appears to be a clear relationship to fire, with a winter burn stimulating an increase in numbers and vigor the following year.

With some clues as to the species' site preferences and management requirements, we have looked for ways to increase our *W. amplexifolia* population and establish new ones. Propagation under nursery conditions yielded an impressive number of vigorous-looking seedlings, all of which succumbed to caterpillars, leaf miners, and fungal pathogens before blooming. Repeated attempts to raise this plant with petunia-growing technology failed to keep a plant alive long enough to flower.

Direct seeding into our natural area (after a burn, on a site where *Warea* had never been seen before) produced slightly better results. Two thousand seeds produced 600 seedlings, of which 16 plants lived to flower and set seed. This high attrition rate (from 600 to 16, although the rest of the 2,000 seeds may come up next year) is not really surprising for an annual, but it is a cautionary tale for those relying on seed storage as a conservation tool. It is apt to take far more seed than we imagine to establish a full-sized population. Only research on a scale large enough to reflect natural systems will provide the answers, of which germination rates are only a small part of the story. It will be several years before we can draw any more conclusions about this *Warea* experiment, although this fall we will seed another area with 10,000 seeds.

In contrast to the above-mentioned examples, however, ex situ preservation has worked spectacularly well for other species in our collection. Woody species like the pygmy fringe tree, *Chionanthus pygmaeus*, the four-petaled pawpaw, *Asimina tetramera*, and the exceptionally rare *Ziziphus celata*, as well as semi-woody perennials like the three scrub mint *Conradina* species -- *C. brevifolia*, *C. glabra*, and *C. grandiflora* — are long-lived with more stable personalities. In each case, the horticultural technology is well enough developed to produce the plants in large numbers and keep the populations alive indefinitely.

Conclusion

And yet, as personally satisfying as a bed of 50 or 100 or even 200 of these rare plants may be, it is still not good enough. In fact, seeing such an ex situ population manifest in a 10' x 25' bed makes it clearer than ever that we have not yet grasped the scale of the problem. We should be talking about thousands of plants on dozens of sites. In most cases, although each species' situation is a little different, real species recovery will require much more than what we are able to preserve of the natural populations and propagate ex situ — and only large-scale introduction projects can accomplish this feat.

We see the information learned from our ex situ propagation research as an important step in what must be a multi-faceted and interdisciplinary approach to rare plant preservation, combining the knowledge and resources of botanists, horticulturalists, land managers and public agencies.

Here in central Florida we are in a desperate last-ditch effort to "save the pieces" after decades of thoughtless and unintelligent tinkering with our environment. Where habitat destruction has been relentless and population numbers have dwindled to pathetic levels, bold action is called for, even — regrettably — with sometimes incomplete knowledge and imperfect technology.

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Down But Not Out: Reintroduction of the Extirpated Malheur Wirelettuce, *Stephanomeria malheurensis*

by
Robert L. Parenti and Edward O. Guerrant, Jr.

INTRODUCTION

Known only from a single site in southeastern Oregon, the herbaceous annual, *Stephanomeria malheurensis* (Malheur wirelettuce) was discovered in 1966, placed on the federal list of endangered species in 1982, and by 1985, had apparently become extinct. Recovery efforts begun in 1987 by the U.S. Fish and Wildlife Service (USFWS), along with the Bureau of Land Management (BLM), were possible only because viable seed had been stored offsite and was available for reintroduction. The seed had been carefully maintained by Dr. Leslie Gottlieb, an evolutionary geneticist at the University of California at Davis, who had discovered, described, and worked extensively with the plant.

Stephanomeria malheurensis is of considerable scientific interest in part because it belongs to one of the most well-studied parent-offspring relationships known between any two species. The self-pollinating *S. malheurensis* is almost certainly a recent derivative of the outbreeding *S. exigua* ssp. *coronaria*. Sympatric to the point of sometimes having intermingling stems, *S. malheurensis* occurs at the northern limit of its progenitor's range, which extends throughout much of California and the west. Malheur wirelettuce, however, is known to occur only at a single site: an area on BLM land about 25 miles south of Burns, Oregon, which is now designated the South Narrows Area of Critical Environmental Concern.

Recovery efforts have been complicated by the fact that the original habitat was altered both by a fire in 1972, and subsequent invasion of cheatgrass (*Bromus tectorum*) — a very aggressive weed native to Europe. Never known to number more than 750 plants, the decline of the wirelettuce was first noted in 1980 -1981, and by 1985-1986, no individuals could be found. Simultaneously, cheatgrass coverage increased dramatically. What had been mostly bare soil was replaced by nearly a 100% cheatgrass cover, suggesting that interspecific competition may have contributed to the decline in *S. malheurensis*. Possible alleopathic effects of cheatgrass are being investigated by the USFWS in cooperation with Boise State University. Preliminary studies show that cheatgrass does inhibit germination, growth, and development of selected species.

Robert Parenti at the field site pointing to where the last naturally occurring wirelettuce plant was found (flags mark seedlings presumed to be offspring) Photo: L.R. McMahan, Berry Botanic Garden

RECOVERY RESEARCH

Experimental Design

Because the ability of the sole original site to support Malheur wirelettuce appears to have been compromised, the recovery plan includes research intended to answer a number of questions about the biology of the species in order to determine what, if any, managerial options exist for the BLM to re-establish and maintain a viable population of Malheur wirelettuce. To better understand the biology of Malheur wirelettuce and its competitive interactions with cheatgrass and other associated species, the BLM, in conjunction with the USFWS, established a series of experimental plots at the original site in 1987.

Four separate plots were established, each surrounded with rodent-proof wire mesh enclosures extending four inches below ground and 32 inches above. Each plot is dominated by one of four species common to the study area: the three natives, green rabbitbrush (*Chrysothamnus vicidiflorus*), big sagebrush (*Artemisia tridentata* var. *wyomingensis*), and Great Basin wildrye (*Elymus cinereus*); and the exotic cheatgrass. Three of the plots are square, 5m on a side, and the cheatgrass plot is twice that size, 5m by 10m. In 1987, all plots except the cheatgrass plot were completely weeded of cheatgrass. Into these plots were planted 1,000 seedlings of *S. malheurensis* seedlings, which had been grown by The Berry Botanic Garden of Portland, Oregon, from seed provided by Gottlieb. The plants were thoroughly moistened on planting and thereafter for about four weeks until the plants had become established.

Survivorship and reproduction were generally high. In 1988, with seed in the soil from plants grown there the year before, the 5m by 5m plots were divided into four equal quadrants, 2.5m on a side. Two of these were left undisturbed and termed "untreated"; two were weeded completely of cheatgrass and referred to as "treated." The cheatgrass-dominant plot was divided into eight paired squares, 2.5m on a side, with four of them remaining undisturbed and four weeded of 50% of the existing cheatgrass.

Preparation of the study plots was similar in the 1989 field season. The wildrye and rabbitbrush plots were again set up with two treated and two untreated quadrants each. The paired-quadrant 50%/100% regime was maintained in the cheatgrass plot. Due to small mammal depredation of Malheur wirelettuce in the sagebrush plot in 1988 — a recognized natural threat to the species — that plot had to be re-established in 1989. It was weeded of all cheatgrass, and 80 greenhouse-established seedlings from The Berry Botanic Garden were planted. Thus, there were no contrasting treated/untreated quadrants in the sagebrush plot in 1989. In 1990, the cheatgrass plots were prepared in the same manner as the other three plots, i.e., portions were divided into "treated" and "untreated" quadrants.

Results

Data collected through 1989 show that *S. malheurensis* plants are influenced differently by the various species. Wildrye appears neither to affect negatively the vigor of Malheur wirelettuce, nor alter its phenology. Rabbitbrush, on the other hand, does appear to have some negative effect. The effects of sagebrush cannot yet be determined because of the herbivory and subsequent necessity to reestablish that population. The most profound effects on Malheur wirelettuce were associated with cheatgrass: plants grow more slowly than in its absence, and the phenological events of flowering, seed set, and seed release are significantly later in plots which include the exotic grass. In addition, survival and recruitment rates are lower in those plots.

Also in 1990, all of the approximately 250 seedlings from The Berry Botanic Garden were planted outside of the plots in order to: 1) ascertain the effects of native herbivores, and 2) establish experimental plots to examine the longevity and germination patterns of naturally occurring soil seed banks. Native herbivores, primarily the blacktailed jackrabbit, destroyed all of the seedlings that were not protected and watered for the soil seed bank experiment. This may not have been a "fair test" because the year was so dry, and therefore these plants were conspicuous forage.

Fruits of Malheur wirelettuce Photo by E. Guerrant

The replicated soil seed bank study is designed to follow the fate of a single cohort of seeds: what proportion of seeds will geminate either in the fall or spring of the first year, and for how many years will seedlings from a single year's seed crop emerge thereafter? Populations of annual plants rely for survival on recruitment from the supply of viable seed in the soil. The pattern of *S. malheurensis* seed survivorship in the soil has taken on more than an academic interest because there appears to have been no seedling recruitment at all in 1990. It was a very dry and cold spring, which apparently did not provide the very narrow temperature window in the upper 50s (degrees Fahrenheit), known to stimulate germination. This may very well have been a blessing, because the rest of the growing season was dry as well. If there remains a viable soil seed bank then this may not be a serious setback. In any case, considerable quantities of stored seed now exist offsite.

CONCLUSION

This ongoing recovery effort, begun with the planting of seedlings in 1987, has been extremely important in establishing research and management directions that must be taken in the future. A focus of continued study is clarification of the effects of native and exotic species on wirelettuce. Data collected over a period of years will be necessary to provide information about their interactions and allow investigators to determine whether changing environmental conditions alter those interactions. Without the experimental plantings, we would have little to go on to determine the survivability of *S. malheurensis*. This project is an excellent example of the cooperation between the USFWS, BLM, academic scientists, and private organizations to enhance the survival of a species with such scientific importance.

The primary goal of continued study is to determine management options for protection and perpetuation of Malheur wirelettuce. In retrospect, the existence of artificially "banked" seeds maintained offsite appears to have made the difference between extinction and continued existence for *S. malheurensis*.

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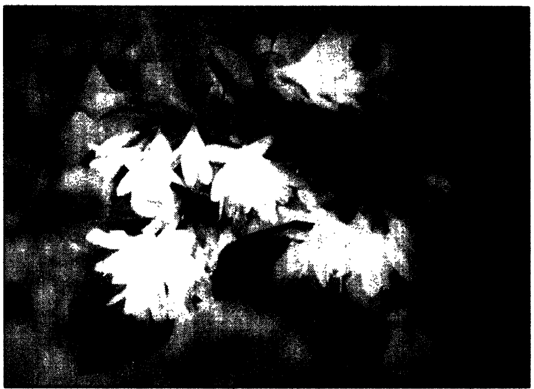
Reintroduction of the Texas Snowbell, (*Styrax texana*)

by
Paul Cox

Few other areas in the United States can boast the ecological diversity found in Texas. A combination of vastly different edaphic and climatic factors give rise to no less than ten popularly recognized vegetational regions. Rainfall across the state varies from as much as 50 inches in the eastern pine hardwood forests to as little as five inches in the western deserts and mountains.



Texas snowbell Photo: San Antonio Botanical Center

One of the most significant areas is the Edwards Uplift, a limestone plateau located in central Texas, locally known as the Texas Hill Country. This fairly rugged region consists of rolling hills deeply cut by steep-sided creeks and ravines. Most of the Hill Country area has been severely abused during the last century. Coupled with a warming and drying trend, overgrazing has devastated much of the natural flora. Not only is the native white-tailed deer population at an all time high, but the proliferation of exotic game animals such as the axis deer and aoudad sheep have added to the plight of native vegetation. The widespread increase of sheep and goats has only exacerbated the degradation. Large areas have been overgrazed to the point that the survivability of some plant species is seriously in question.

Enter here the Texas snowbell (*Styrax texana*) located deep within the heart of the Hill Country. This obscure shrub grows exclusively on cliffs associated with the deeply cut waterways of the area. Due to low population numbers, the species was listed as federally endangered in 1984. Today there are currently fewer than 40 individuals known from seven widely scattered populations. Furthermore, some of these populations consist of only a single specimen.

The Texas snowbell is actually quite an ornamental shrub. Away from the steep cliffs it stands about ten feet tall with handsome heart-shaped leaves that are dark green above and whitish below. In mid to late April the plants are decorated with beautiful 1/2 to 3/4 inch-long, bell-shaped white flowers. These are followed later by 1/4 to 1/2 inch hard, round fruits usually containing a single seed. Unfortunately for the fruits, the parent plants are almost always found leaning out from cliffs where the seeds fall into creeks or dry streambeds and are swept away by the next good rainfall. This situation, together with the fact that the plants are eagerly eaten by herbivores, has combined to push the Texas snowbell to the brink of extinction.

Captive Propagation

The Texas snowbell has long been recognized as Texas' most seriously threatened native plant species. Pioneer propagation work on this and many other endangered Texas species was performed by the now defunct Rare Plant Study Center (RPSC) of the University of Texas at Austin. In 1985, sometime after the demise of the RPSC, the San Antonio Botanical Center became a participating institution with the Center for Plant Conservation (CPC), an organization dedicated to preserving our nation's endangered flora. One of the CPC's goals is to establish secure captive populations as insurance for some of the more critically endangered species. The first plant brought into the San Antonio Botanical Center's collection under this program was the Texas snowbell.

Building upon the early groundwork laid by the RPSC, we soon found that the Texas snowbell took to cultivation with only a few minor problems. Thus it did not take long to develop a fair-sized captive population of about 150 plants. We determined that the best germination rate resulted from stratifying the seed for 60 days in moist sphagnum. By satisfying these production and subsequent cultivation requirements, we met our first goal of establishing a captive population.

Reintroduction Efforts

Encouraged by this success, we began to explore the possibility of reintroducing these plants into protected habitats in the wild. An Institute of Museum Services grant helped pave the way for this endeavor, but the project developed in earnest when Charlie MacDonald, with the U.S. Fish and Wildlife Service (USFWS), approached us and proposed a three-year grant to fund Texas snowbell reintroduction efforts.

In order to carry out this task it was our good fortune to team up with Toney Keeney, a professor at Southwest Texas Junior College in Uvalde. Toney has spent many years studying the snowbell, knows the populations thoroughly, and long ago gained the trust of the sometimes wary landowners. Rather than resent the interference from outsiders, Toney welcomed us with open arms. The reintroduction of the Texas snowbell was something he had long contemplated but lacked the resources to implement. He had already picked out a site and garnered support of one landowner and the ranch foreman. Gaining such support can be a difficult task as rural Texans are often intensely resentful of programs which they view as government interference. However, for this job Toney Keeney is well-suited, with a polite laid-back and sincere approach that is very disarming. A second landowner was secured and the project was ready for implementation. The effectiveness of these efforts illustrates the importance of working closely with local authorities who are invaluable to such a complex project.

Overall, the selected reintroduction sites are good reflections of actual snowbell habitat, i.e., steep banks near reliable water. As much as was possible under the circumstances, careful consideration was given to every important aspect of site selection. The plants had to be confined to areas within their historic range, and kept as far as possible from populations of the closely related, somewhat more eastern sycamore-leaf snowbell (*Styrax platanifolia*) to prevent the possibility of hybridization between these two species whose ranges do not normally overlap. Other important aspects taken into account include available moisture, soil type, exposure, and similar associated plant species. We eventually chose two reintroduction sites, called Mare Creek and Cypress Creek.

Planting the progeny had to be performed while perched upon eight foot ladders carried into the field. Once installed in their new homes on the bluffs adjoining the creek, the plants had to be watered and measured regularly throughout their early years. This was heroically carried out by Toney Keeney and recruits, who had to use the same precarious procedures as in the planting expeditions. To protect the young plants from marauding herbivores and rodents, the seedlings were covered by small chicken wire cages. Toney's ongoing maintenance and record keeping is entirely funded by the USFWS grant.

In 1987, we planted a total of 25 seedlings at the Mare Creek location and 24 at the Cypress Creek site. The Mare Creek seedlings, planted on May 21, averaged approximately 14 cm in height, while Cypress Creek seedlings, installed on July 13, averaged 16 cm.

As fate would have it, during the last week in May and early June, record rainfall fell on the Mare Creek location. Most of the plants there were either destroyed or damaged by flood waters and associated debris. Consequently, on July 17, 24 replacements were replanted, but on the limestone ledge above Mare Creek. The Cypress Creek "population," on the other hand, was unaffected by the heavy rains, although one plant was lost to drought stress during the hot summer months. Many plants, at both locations, began to put on new growth when the cooler fall weather arrived in the fall.

After three years the project results have been very encouraging. The Mare Creek site has 18 survivors that now average 43.2 cm in height. To accommodate this growth, new cages were constructed for eight plants and installed in February, 1990. At Cypress Creek, mortality was lower but growth was slower. Only one of the original 24 plants was lost and the remainder have added more than 24 cm in height. While they are not growing fast, these plants look healthy and appear in stable condition. This phase of the project thus seems to be fairly successful. The protected plants are growing satisfactorily within the confines of their wire cages, although they have yet to reach breeding size.

The ability of these plants to regenerate under the present conditions is questionable. The bulk of the Hill Country is so seriously overgrazed by both native and exotic herbivores that protection is necessary on a community level, not just for individual plants. Recent work by the Texas Parks and Wildlife Department found that a fenced population of Texas snowbell demonstrated successful preliminary regeneration.

These combined efforts indicate two things. First, the Texas snowbell can be reintroduced into protected native habitats, provided it is defended against animal predation. While the project offers secure individuals, it must be questioned whether or not these individuals can become a successful regenerating population under present circumstances. Herbivorous animals exert a great pressure on virtually all Hill Country plants; the Texas snowbell is noteworthy only because it already had such relatively low population numbers.

Planting *Styrax texana* on limestone bluffs at the Mare Creek site Photo by C. McDonald

Secondly, the work by Texas Parks and Wildlife has shown that protected populations seem to be capable of sexual reproduction, but only within the confines of their enclosure. However, there is, at present, insufficient funding or enthusiasm for a project that would exempt herbivores from an area large enough to accommodate the need for an expanding Texas snowbell population.

Conclusion

Our part of the project has been largely successful to date. Propagation and cultivation techniques are refined to the point that the Texas snowbell can be grown in captivity with confidence. Reintroduced plants have survived the crucial first few years of establishment and are growing well.

Although the snowbell may not actually be the most threatened plant species in Texas, it is probably the most well-known and longest-studied of our natives. Preliminary results are encouraging. With increased education and public awareness, success of the project may prove to be a turning point in the management of Texas endangered species.

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Transplantation of an Otherwise Doomed Population of Barrett's Penstemon, *Penstemon barrettiae*

by
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Unlike other projects reviewed in this Special Issue, this work does not address a formal recovery effort for a listed species. Rather, it describes the first phase of an opportunistic experiment involving the transplantation of rooted cuttings taken from a newly discovered but doomed population of *Penstemon barrettiae*, a candidate (C2) for listing, to a nearby site. (Candidate 2 species are imperiled species deemed by the U.S. Fish and Wildlife Service to have insufficient data for listing.)

With large bluish-green leaves and showy tubular rose-colored flowers, the small shrub, *Penstemon barrettiae*, is an attractive plant. Until this population was discovered in 1985, the approximately dozen known populations of the species were limited to about a five mile stretch along both banks of the Columbia River in Washington about 50 miles east of Bonneville dam, with scattered populations to the north along the Klickitat River. It is a colonizing species that grows on rocky cliffs and gravelly hillsides, often along highways and railroads where it is vulnerable to indiscriminate collecting and herbicide use.

This disjunct population was discovered shortly before the Army Corps of Engineers was scheduled to begin construction of a new navigational lock at the Bonneville Dam. Although at the time not legally required to protect this "Candidate 2" species (even though its habitat was slated for certain destruction), the Corps helped support a cooperative project to salvage cuttings, and facilitated their transplantation at two nearby sites. While wholesale transplantation of endangered populations is clearly not the method of choice to preserve genetic diversity, in this case the alternative was certain destruction.

Translocation Efforts

In May 1986, Ms. Julie Kierstead, then of The Berry Botanic Garden, took multiple cuttings of vegetative shoots from as many plants as she could reach — an amount equalling somewhat less than half the plants visible — either from the ground or in a "cherry picker" supplied by the Corps. The cuttings were taken to the Garden where an attempt was made to root them. Results were generally encouraging, but varied widely from complete success with some clones (9 of 26 clones), to complete failure in others (3 of 26 clones). The term "clone" is used here to refer to all cuttings taken from a single plant, although not all separate plants are necessarily different genetic individuals because the plants can spread naturally by the rooting of shoot fragments.

An initial planting in the spring of 1987, which was undertaken in a conspicuous location near the visitor center for publicity purposes, was done at the wrong time of year. Consequently, this planting resulted in 100% mortality the first summer. However, in the spring of 1988, 70 rooted cuttings from 21 different parent plants were planted in two more appropriate habitats nearby, both already occupied by *P. barrettiae*. As of May 1990, at least half of these plants were still alive (some loose tags were

"While wholesale transplantation of endangered populations is clearly not the method of choice to preserve genetic diversity, in this case the alternative was certain destruction."

Ed Guerrant and Jack Poff inspecting a site for introduction of *Penstemon barrettiae* at Bonneville Dam.
Photo by L.R. McMahan

found in the most recent survey, which may belong to surviving plants), and three are beginning to produce flowers. The existing shoots of several plants have largely died back, but they are producing vigorous new shoots from the base of the plant. Of the 21 clones planted out originally, 15 (71%) are still represented by at least one living individual. Much of the mortality appears to be due to soil erosion.

> **"From a strategic or policy perspective, another weakness [of translocating populations] is the apparent ease of the method; where land usses conflict, simply moving the offending occupants to another suitable site should not be offerred as an early or preferred option."**

Evaluation of Strategy

Simply moving a threatened population by transplanting cuttings superficially offers an appealing, pragmatic option for conserving genetic diversity —at least in the short term. However, the method involves many uncertainties and may not represent a generally desirable alternative. The most obvious shortcoming is illustrated by the fact that all plants rapidly died in the first planting because the habitat was apparently not adequate, even for this easily propagated colonizer. The limited number of "surplus" individuals in rare populations makes such failed experiments potentially disastrous for the survival of an imperiled species.

However, failure also can be protracted over a much longer time span, despite an initial appearance of success. In this case, success has been achieved thus far only in locations already occupied by the species — a situation not always available, and not necessarily even desirable. All of the plants originally at Bonneville before construction may well have been one breeding population anyway, so moving "aliens" into an already occupied site (the only suitable habitat available nearby) probably did not contaminate the indigenous gene pool. But genetic contamination remains an important unanswered question.

There are other weaknesses as well: genetic variation may well be lost, and success cannot be determined for a considerable length of time. Not all plants yielded viable cuttings, and not all clones show equal survivorship. Consequently, the genetic composition of the resulting population is probably different and less diverse than the original population. From a strategic or policy perspective, another weakness is the apparent ease of the method; where land uses conflict, simply moving the offending occupants to another suitable site should not be offered as an early or preferred option. Another problem with transplantation is that it may take many years before we know if even this remarkably straightforward attempt has succeeded.

> **"This [translocation] project is, I believe, best viewed as an opportunistic experiment — perhaps a preview of things to come if global warming occurs."**

A strength of this strategy of translocating threatened populations is that ideally, given enough lead time before a population's destruction, all of the genetic information in the population can be salvaged—at least temporarily. This information can then be housed at several places at once to minimize the chance of mortality. Even in this nearly ideal situation, though, some clones were lost. Nevertheless, the Garden still has some of these clones, and additional outplanting is planned in the future. Seeds were also collected along with the original cuttings, and are currently housed in The Berry Botanic Garden's Seed Bank for Rare and Endangered Plants of the Pacific Northwest. These offer further options for future planting.

Ultimately, when construction on the new lock is finished, there will be new apparently suitable habitat available, and ideally we will attempt the ultimate goal of this project: to recolonize the unoccupied cliff face habitat with the original occupants, or their progeny. This project is, I believe, best viewed as an opportunistic experiment—perhaps a preview of things to come if global warming occurs. It is too early really to evaluate the results with any confidence, other than to say that some genetic diversity has apparently been lost, but that some remains where otherwise there might have been none.

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Colorado Squawfish Reintroduction Efforts in the Lower Colorado River Basin

by
Buddy L. Jensen

INTRODUCTION

Dam and irrigation diversion construction have extensively modified most major U.S. rivers. Additionally, rivers and streams commonly were treated with fish toxicants when projects were completed in order to remove "trash" fish prior to dam closure. Most native fishes were considered expendable in the interest of carrying out these development projects or enhancing sport fishing programs. These projects, when combined with other habitat modifying activities, i.e., pumping of ground water, overgrazing, and poor logging practices, had a deleterious effect on the natural quality of our river systems and their fauna. Non-native sport fish species were stocked extensively with little or no consideration given to potential impacts on native fishes or other aquatic organisms. Unfortunately, these activities have resulted in the decline of many native fishes throughout the country. This paper discusses the demise of, and some of the recovery efforts for, one of the unique native fishes impacted by development activities in the Colorado River basin of the Southwest.

The rivers and springs of the American Southwest, along with their native aquatic fauna, changed dramatically during the 20th century (Miller 1961). Over time, the activities described above resulted in a continuous and progressive decline in the distribution and abundance of the unique endemic ichthyofauna of the region (Johnson and Rinne 1982). The voices raised in opposition to these practices were a select few, those with insights far beyond the majority of fishery managers of the time. Twenty-two years ago in Death Valley, some of these foresighted individuals were instrumental in establishing the Desert Fishes Council — a group of concerned professionals whose primary goal was to conserve, protect, and perpetuate native desert fishes and their habitats. Twelve years ago, the combined efforts of the Desert Fishes Council, the academic fisheries community, and key U.S. Fish and Wildlife Service (USFWS) personnel led to the changing of the role of the Dexter National Fish Hatchery and Technology Center (NFHTC) in Dexter, New Mexico, from rearing sport fish to holding, studying and culturing imperiled native fishes of the Southwest. This was a major step forward in the effort to conserve native fish species.

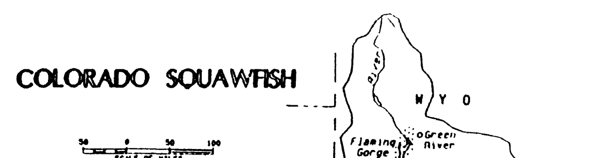


Figure 1. Past and present distribution of the Colorado squawfish in the Colorado River system (from Colorado Squawfish Recovery Plan 1990).

Dexter NFHTC serves as a refuge for imperiled fishes, provides opportunities for research and development projects using captive-reared fish, and produces selected species for recovery programs identified in recovery plans or other interagency documents (Jensen 1983, Rinne et al. 1986). The Dexter NFHTC currently maintains 14 species of imperiled fishes. Since the inception of the current program, more than 25 species have been held at Dexter for one or more purposes. The Colorado squawfish, *Ptychocheilus lucius*, a federally listed endangered species (Federal Register [32(43):40001] March 11, 1967 and [39(3):1175] January 4, 1974) is one of the species being maintained and produced at Dexter NFHTC for reintroduction purposes. This account briefly examines the captive breeding program for Colorado squawfish at Dexter NFHTC, and reintroduction efforts for the species in the lower Colorado River basin in Arizona.

DESCRIPTION AND CURRENT RANGE

The Colorado squawfish is North America's largest, native minnow. The term "*lucius*" means "pike-like," and squawfish do resemble northern pike in body form: individuals are elongate, compressed dorso-ventrally, and have a long flattened head. Adults have a dark olivaceous-colored back which lightens along the sides to a whitish belly. Although a true minnow (Family Cyprinidae), it historically attained lengths and weights approaching six feet (183 cm) and 100 pounds (45 kg) (Miller 1961). In recent years, however, individuals over 30 inches (76 cm) and 15 pounds (6.8 kg) have been difficult to find.

Like the other big native fishes of the Colorado River -- razorback sucker (*Xyrauchen texanus*), bonytail chub (*Gila elegans*), and humpback chub (*Gila cypha*) -- Colorado squawfish are long-lived. However, squawfish grow and achieve sexual maturity much slower than the other three species; captive-reared razorback sucker and bonytail chub can reproduce at two years of age, while the squawfish must be six years old (Jensen 1983, Rinne et. al. 1986). Recent studies of adult bonytail chub and razorback sucker revealed that both species live for more than 40 years (McCarthy and Minckley 1987, Minckley et. al. 1989). Although data are limited, this author estimates that historically, Colorado squawfish lived to be 60 to 80 years of age and perhaps older. Sixteen year-old squawfish maintained at Dexter NFHTC average about 25 inches (63.5 cm) in length and 5.7 pounds (2.6 kg) in weight. Thus it is easy to speculate that a six-foot, 100-pound fish would be extremely old, perhaps over 100 years old.

Native only to the expansive Colorado River basin of the western United States and northwestern Mexico, the "white salmon of the Colorado" was well known by early settlers for its white, flaky, sweet meat and its migratory spawning habits (Minckley 1973). This voracious predator was historically common in the main river channels throughout the Colorado River basin (Figure 1). An indication of their former abundance was noted by Miller (1961) who stated, "Until about 1911, the species was so abundant in the lower Colorado that individuals got into the irrigation ditches and were pitchforked out onto the banks by the hundreds for use as fertilizer."

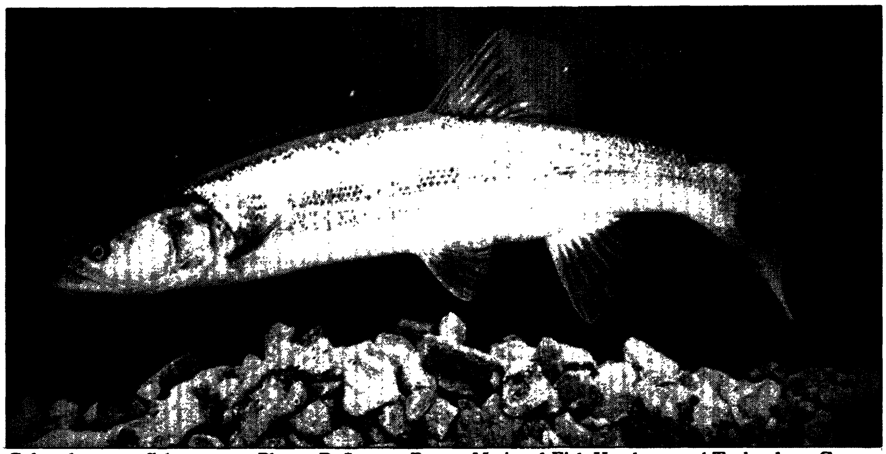
SPECIES DECLINE

The decline of the Colorado squawfish is probably related to a combination of factors, the most important being: 1) direct loss of habitat; 2) blockage of spawning and seasonal migration routes by dams and diversions; 3) changes in flow and temperature regimens below dams; and 4) interactions with non-native, introduced fishes.

Historically, the Colorado River was savage and unpredictable, known simply by its one dominating color — el Rio Colorado, the Red River of the West (U.S. Fish and Wildlife Service 1990). Seventeen hundred miles (2,700 km) long and dropping more than 2.0 miles (3.2 km) in its journey from the high mountains of Wyoming, Utah and Colorado to the Sea of Cortez, the river courses through high canyons 90% of the way. In this turbulent, silty river evolved one of the most remarkable assemblages of American freshwater fishes, uniquely adapted to survive some of the most turbulent aquatic conditions on earth.

Beginning with the construction of Roosevelt Dam on the Salt River in 1913, and continuing with more than 20 additional mainstream and tributary structures completed since then, humans have dramatically altered the character of el Rio Colorado. No longer flowing freely to the ocean, the River has become a series of large impoundments over much of its course, connected by cold, clear waters uncharacteristic of the former river. Some of the Arizona tributaries are now dry in their lower reaches. For example, over 200 miles (320 km) of the Gila River flows only seasonally during intense summer thunderstorms or in the spring following uncommonly wet winters.

These disruptions to the physical riverine environment, coupled with introduction of numerous exotic fishes, have caused a steady decline in the abundance of the native fish fauna. There are 35 fish species native to the Colorado River, but more than 50 introduced fishes have become established (W.L. Minckley, pers. comm.). Perhaps more importantly, exotics comprise an estimated 99% of the total fish biomass present in the system. In addition to being greatly reduced in numbers, the Colorado squawfish now occupies only about 25% of its former range, and has been extirpated from the entire lower basin (Figure 1).



Colorado squawfish Photo: B. Jensen, Dexter National Fish Hatchery and Technology Center

RECOVERY ACTIVITIES

Because of existing conditions, recovery of the Colorado squawfish will require a major effort. Between 1973 and 1979, 59 individuals were collected from the wild and transferred to Willow Beach NFH, Arizona. From these fish, several year classes were produced (Toney 1974; Hamman 1980, 1981). Representatives of two of these classes (1974 and 1981) are currently held at Dexter NFHTC. The 1974 year class has been spawned for ten consecutive years to produce young-of-the-year for research and reintroduction purposes. Genetic diversity of the captive broodstocks was maximized to ensure genetically fit offspring for reintroduction (Ammerman & Morizot 1989).

Spawning and Culture

Willow Beach and Dexter hatchery managers developed and refined squawfish spawning techniques. To induce ovulation, females receive an intraperitoneal injection of carp pituitary as they approach final maturation of ova (Hamman 1981, 1986; Inslee 1983). Egg production averages about 18,000/lb (39,690/kg) of body weight; one ten-pound female at Dexter NFHTC gave over 200,000 eggs during a single spawn.

Ovulation occurs 18 to 24 hours following injection. Ovulated eggs are hand-stripped into porcelain pans containing sperm diluent solution. Milt from two or more males is expressed into the spawning pans directly from the fish. Water is then added to the mixture to activate the milt. During the spawning process, the entire mixture is stirred with a feather to ensure adequate mixing of sexual products, and to prevent the eggs, which are highly adhesive, from sticking to the pan. Following fertilization, a slurried bentonite clay solution is added to the mixture to reduce the adhesiveness of the eggs prior to placing them in floating egg baskets. The eggs are washed gently to remove the clay solution and allowed to water-harden for 30 minutes prior to enumeration.

Eggs are enumerated gravimetrically and placed in incubators receiving a flow of aerated, 70° Fahrenheit water. Hatching begins at about 96 hours and peaks at about 108 hours. Sac fry are transferred to tanks where swimup occurs at approximately 96 hours post hatching. Swimup fry are stocked in earthen rearing ponds at a density of 125,000/surface acre. Swimup fry are also shipped live in plastic bags containing water and oxygen, via air freight, to other cooperating agencies and institutions. Colorado squawfish attain a length of three inches (7.6 cm) during their first growing season (June - October) at Dexter NFHTC. Fingerlings and adults are transported in standard fish distribution units.

Stocking

Protecting surviving wild stocks of Colorado squawfish is important, so reintroduction efforts have been approached differently between the upper basin states (Wyoming, Utah and Colorado) where wild populations still exist, and the lower basin states (New Mexico, Arizona, Nevada and California) where Colorado squawfish were extirpated at least 25 years ago (Minckley and Deacon 1968). The primary objectives guiding early reintroduction efforts in the lower basin included attempts to locate suitable habitat for reintroduction, setting up stocking protocols for a listed species, and conducting post-stocking dispersal and survival studies. Since extant populations must be included in upper basin strategies, their recovery/reintroduction efforts have progressed more slowly.

Reintroduction and monitoring of Colorado squawfish in the Verde and Salt River systems of central Arizona began in 1985 (Johnson 1985). Individuals stocked and populations established under this recovery effort are classified as "nonessential experimental" under Section 10(j) of the Endangered Species Act, as amended (Federal Register Vol. 50(142): 30188-30195) (Brooks 1986). (Nonessential populations are not considered to be essential to the survival of the species and thus are not afforded protection under Section 7 of the Act.) This approach to reintroduction was necessary because water development and consumptive-use interests feared that water use might be restricted or controlled due to the presence of a federally protected species.


Aerial view of Dexter National Fish Hatchery and Technology Center
Photo: B.Jensen/DNFHTC

Since 1985, a total of 488,117 Colorado squawfish fingerlings from Dexter NFHTC have been stocked into central Arizona waters, along with an additional 301 adult, surplus broodfish. The Dexter facility also provided thousands of fry and fingerlings to Page Springs State Fish Hatchery in Cornville, Arizona, for rearing and stocking. Between the two stations, a total of 518,258 fingerling or adults have been reintroduced into historic habitats in the lower basin. Although these numbers seem significant, they number less than the ova from three ten-pound females, and are probably infinitely small compared to the numbers of young fish produced under historic conditions.

Monitoring

Within the constraints of personnel, funding, and time constraints, biologists from the Arizona Game and Fish Department, USFWS, and Arizona State University have evaluated the success of introductions annually. River reaches receiving reintroduced squawfish have been monitored up to three days immediately following stocking to evaluate post-stocking dispersal of, and predation on, stocked fish. Generally, the stocked fish move rapidly downstream and are preyed on by both native and exotic predators.

Additional monitoring has been carried out a few weeks to six months following stocking. Based on information collected during the latter surveys, survival of introduced squawfish appears low. Recovery of stocked fish has occurred on numerous occasions, but primarily within a few weeks following stocking; only a few individuals that have been in the system for more than six months have been recovered. Although results are inconclusive because of difficulty in sampling the large, remote river habitats, they are nevertheless indicative of the situation, and support the premise that successful recovery of Colorado squawfish will occur neither quickly nor easily.

SUMMARY AND DISCUSSION

The progressive aspect of the Colorado squawfish reintroduction program has been the successful development of spawning and culture techniques for the species; large quantities of fish can be produced for reintroduction purposes upon request. The difficulties encountered have been in the attempts to successfully reintroduce fingerling fish.

The reasons why reintroductions have been minimally successful cannot be precisely determined, but probably include a combination of some or all of the following unranked factors: 1) dams and other habitat alterations; 2) presence of numerous exotic species that interact with and prey on reintroduced fish; 3) native species predator/competition interactions brought about by altered habitats; 4) reintroduction of comparably small numbers of fish in large, diverse habitats; 5) sampling inefficiencies due to personnel and habitat constraints; 6) life history parameters (Colorado squawfish are long lived and grow slowly, so one cannot expect immediate results from the efforts made thus far); and 7) homing.

Tyus (1985) and Tyus et al. (1985) reported homing behavior and spawning migrations for wild Colorado squawfish in the upper basin. Tyus cautioned about stocking hatchery-reared fish into habitats containing wild fish because their inability to home could possibly impact extant populations of the species. If Tyus's conclusions are correct, this could also be a problem in the lower basin even though natural populations have already been extirpated. If squawfish imprint during early life stages, migrate back to spawning grounds to reproduce, and will spawn only at those locations, hatchery-reared fish likely will not reproduce in the wild since they imprinted at the hatchery which is out of basin. However, observations made at Dexter NFHTC suggest that squawfish do not necessarily require spawning migrations and homing to successfully reproduce: fish hatched and reared in the Colorado River basin have spontaneously reproduced in ponds at Dexter NFHTC. Although recruitment has been low, it is speculated that it would be much higher if adequate substrate and current were provided to simulate a natural spawning bed. This does not mean that spawning migrations and homing are not important to the continued survival of wild individuals, only that successful reproduction can occur in the absence of these factors. It remains obvious that much more work is needed to fully understand the role that homing and migration play in the life history of this species.

Although the physical presence of dams must have played a major role in the demise of the Colorado squawfish throughout its historic range, they may not be directly affecting the outcome of early reintroduction efforts in the lower basin since all stockings are taking place in riverine habitats upstream from existing structures. Instead, the most apparent problem affecting successful reintroduction is the presence of exotic fishes which both prey upon and compete with squawfish for available food and space resources. Reservoirs impounded by dams harbor large populations of exotic species, thus providing a constant source of fish which move upstream to occupy habitat needed and historically utilized by native fishes. Additionally, non-native fishes continue to be stocked directly into the rivers. Yet, the physical riverine habitat above mainstream reservoirs appears adequate to support the Colorado squawfish and other extirpated species as well. Assuming that existing habitat conditions are maintained in these areas, exotic fishes appear to represent the principal roadblock to successful establishment of Colorado squawfish in the lower Colorado River Basin. Establishment alone does not equal recovery, however, and once established, the role of migration and homing may become important factors in the maintenance of self-sustaining populations.

Other efforts that would enhance recovery possibilities of Colorado squawfish in the lower basin include the following ideas: 1) renovation of river reaches to remove or at least reduce numbers of non-native species, 2) mechanical removal (electrofishing, etc.) of non-native fishes, 3) strict baitfish laws, 4) commercial harvest of riverine catfish populations, 5) liberalization of bag limits for sport fishes in sections of selected riverine habitats, 6) dedication of one or more rivers or streams to native fish only, and 7) public education. Whether politics will ever permit the manipulation of large river reaches to remove non-native fishes in order to re-establish extirpated native fishes, remains to be seen. The Colorado squawfish perhaps represents a unique opportunity in this area since both the states of Arizona and Colorado would like to establish this species as a sportfish with controlled take. Although this designation would do little to emphasize the need to recover and conserve a species, it may be a step toward public acceptance and support of the efforts needed to recovery this and other important native fish species.

Obviously, much work remains to be done, and the task that lies ahead is difficult. A continued major commitment to recovery of the species is required if it is to be saved in the upper basin and successfully reintroduced in the lower basin. Time alone will reveal the final chapter in the saga of the Colorado squawfish — the native white salmon of el Rio Colorado.

ACKNOWLEDGMENTS

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A Fish Faunal Conservation Program: The Lake Victoria Cichlids

by
Les Kaufman

Introduction

In general, efforts to protect the numerous species of imperiled fish in the United States and in the world lag far behind initiatives to preserve furry and feathered "charismatic megavertebrates." For instance, only one marine fish species — the totoaba (*Cynoscion macdonaldi*) — is even listed for protection under the Endangered Species Act. The 1988 IUCN Red Data Book lists 596 imperiled freshwater fish species, nearly half of them from a single body of water -- Lake Victoria in East Africa, bordered by Kenya, Uganda, and Tanzania. Lake Victoria represents the first time an entire faunal assemblage has been listed as endangered by the IUCN, although convincing arguments could be made to list many other assemblages, such as most desert fishes of North and Central America, the stream fishes of the southern Appalachians, the ephemeral killifishes of eastern South America, the orestines of the Andes, the crater lake fishes of the Cameroons, and the rainforest fish faunas of Madagascar, Sri Lanka, and Australia. A complete list of threatened and endangered fishes, based only on presently available information and inference, would probably include more than 10% of the 22,000 or so known fishes.

"Lake Victoria represents the first time an entire faunal assemblage has been listed as endangered by the IUCN, although convincing arguments could be made to list many other[s] . . . A complete list of threatened and endangered fishes. . . would probably include more than 10% of the 22,000 or so known fishes."

In recognition of this crisis, and the limited but important role that captive propagation might play in forestalling it, in 1987 an IUCN Captive Breeding Specialist Group (CBSG) was established to develop propagation and educational programs for endangered aquatic species, especially fishes (see Kaufman 1987, 1989; Brown 1987). Three fish faunas were chosen for Species Survival Plan (SSP) programs: the Lake Victoria haplochromines, Appalachian stream fishes, and North and Central American desert fishes. Of these, the Lake Victoria SSP has progressed the farthest to date.

The restoration and preservation of a single endangered species is a monumental task, thus proposals to rescue entire faunas are particularly suspect. Like invertebrates and plants, fishes are so speciose that only a few fragments of a few faunas could possibly be maintained under an SSP approach using available space and resources. However, even if wise decisions could be made concerning which of the many species to preserve, there are numerous challenges to first overcome in animal husbandry, and in the scientific study of genetics, epigenetics, microevolution, and developmental biology. The Lake Victoria haplochromines provide a potent case history of the nature of these many and complex challenges.

Lake Victoria Haplochromines

Lake Victoria, the world's largest lake, contains an endemic radiation of over 350 haplochromine cichlid fish species which are known for their remarkable diversity of feeding mechanisms and corresponding morphological and behavioral specializations. In addition, the Lake is inhabited by two endemic and three introduced tilapiine cichlids, along with about 40 other fish species from nine other families. The primary cause of the Victorian cichlids' precipitous decline is the successful introduction of a large alien predator, the Nile perch (*Lates niloticus*), although deoxygenation and overfishing have certainly contributed to cichlid decline and failure

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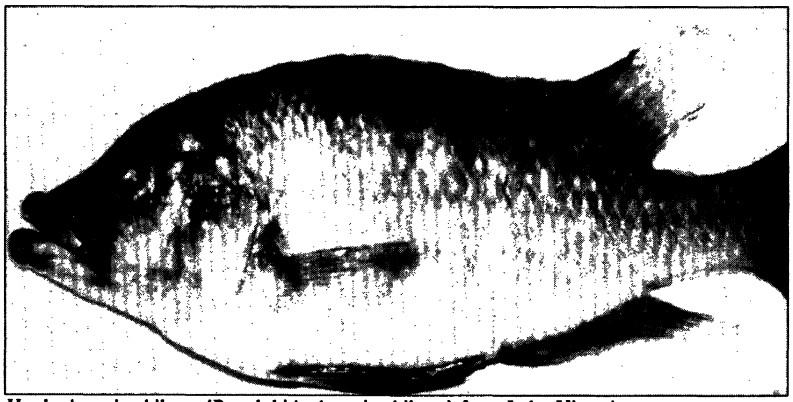
to rebound in areas where Nile perch themselves have been overfished. The actual rate and severity of cichlid extinction is difficult to ascertain, as data is being collected much more slowly than the ecosystem is changing. Damage to fish fauna in certain areas is obvious, albeit complicated: haplochromines are nearly gone from some areas, nile perch from others, perch and haplochromines live together in some places where fishing is off limits, and neither is doing very well where fishing pressure is high. Despite initial skepticism regarding original reports that a mass extinction was occurring, it now appears that these reports were not exaggerated; at least 40% of the fauna is extinct (Lake Victoria Research Team in prep.).

Significance of the Cichlids

The endemic cichlids have far-reaching significance as a component of the Lake Victoria ecosystem. For the people of the region they were a direct and indirect source of protein: native tilapiine cichlids are still the most desirable food fish in the lake; and the introduced nile perch, now the most important fishery, depended on haplochromines for food. Aside from their food value, many of the endemic haplochromines feed upon snails that serve as vectors for schistosomiasis. The elimination of these fish species is thus speculated to have potentially severe impacts on Victorian riparian peoples (Slootweg 1986, 1987).

From a scientific standpoint, Lake Victoria's fish fauna represents a classic example of "explosive speciation" — of particular interest to evolutionary scientists. However, cichlid radiations are characteristic of lakes in the region, and within Lake Malawi and Tanganyika are even comparable in terms of species richness, rates of endemism, and taxonomic composition (Echelle and Kornfield 1984, Greenwood 1984). Thus, some argue that given the similarity in the three faunas, it would be better to focus limited resources for research and conservation upon those faunas which are still reasonably intact and undisrupted. This view is untenable, however, as the faunas are largely independent of one another, and thus offer a replicated evolutionary experiment; the loss of one lake would greatly weaken the strength of evolutionary hypothesis-testing (Kaufman 1989, Avise 1990, Meyer et al. 1990).



Haplochromis chilotes (Paralabidochromis chilotes) from Lake Victoria
Photo by L. Kaufman

"... there are serious ethical implications about allowing the at least partially preventable extinction of several hundred species. Lake Victoria's endemic fishes, and history's judgement concerning human response to their plight, will unquestionably become an important case study in environmental ethics."

Perhaps more importantly, however, there are serious ethical implications about allowing the at least partially preventable extinction of several hundred species. Lake Victoria's endemic fishes, and history's judgement concerning human response to their plight, will unquestionably become an important case study in environmental ethics.

Systematic Characterization of the Fauna

The complexity of haplochromine relationships has been a major obstacle to understanding their evolution and focusing species-oriented conservation efforts. Distinguishing individual species within major species groups is very difficult, and requires comprehensive, carefully curated specimen collections such as exist only at the University of Leiden, the British Museum of Natural History in London, and to a lesser extent, Harvard University in Massachusetts. Many species are still undescribed and may remain so long after they are extinct in the wild.

The coloration of living, sexually active males is one of the most important taxonomic characters for distinguishing haplochromines (e.g., Hoogerhoud et al. 1983), but these colors change when a specimen is preserved. Consequently, researchers have begun to develop color transparencies to accompany specimens. However, there are perhaps only five to six experts in the world who are currently qualified to identify large numbers of species from color transparencies; proper training of new investigators depends upon the efforts of these individuals to catalog a thorough collection.

Although the color of preserved specimens is a useful identification tool, color patterns of individuals within a species vary widely. To further complicate matters, observations indicate that coloration of sexually active males changes ontogenetically (i.e., during development). Molecular systematics, while showing promise as an identification technique, has not provided the hoped-for panacea; electrophoresis often fails to resolve forms that differ markedly in reproductive coloration, ecology, and form. Although progress is being made in the use of mitochondrial DNA (Meyer et al. 1990), this technique is not yet reliable in distinguishing haplochromine species.

Phenotypic Plasticity and Artificial Selection

One of the most puzzling aspects of evolutionary differentiation in tooth and jaw form of haplochromines is that, despite their morphological specializations, many of these species also exhibit a remarkable degree of plasticity in the same characters. For example, the morphology of *Astatoreochromis alluaudi* varies between snail-eating populations, which exhibit massive pharyngeal jaws with molariform teeth, and non-snail eaters, which have proportionately smaller, more gracile jaws with papilliform dentition. Wild-caught individuals of this species exhibited hypertrophied jaws and teeth characteristic of the Lake Victoria population, but their progeny, raised principally on soft foods, exhibited the hypertrophic condition, as did wild-caught individuals maintained on soft foods for several months. Hoogerhoud has since confirmed epigenetic plasticity in this species through elegant experiments and functional analyses (Hoogerhoud 1984, 1986).

In a strict Darwinian sense, acquired characters are not heritable, and therefore such phenotypic plasticity should not pose problems for captive breeding programs. Unfortunately, certain individuals are often better suited, for one reason or another, to the aquarium environment, and thus grow faster, monopolize breeding encounters, and are, by accident or design, selected for subsequent breeding or exhibition. The extreme plasticity of haplochromines ensures that at least some individuals will deviate from natural patterns of trophic development. Since these individuals grow faster than their brethren, they may be preferentially selected, leading to the eventual canalization or "assimilation" of characters that were originally acquired on an individual basis, through epigenesis (i.e., a Baldwin effect). Consequently, it is extremely important to assess the criteria by which individuals are chosen for breeding, as well as the possible need for a captive environment that will foster the development of wild phenotypes. Cryopreservation of the bulk of stock, with genes funneled through a small number of breeding adults, may be the best means of managing haplochromines.

Preserving Species Integrity

Establishing the correct identity and relationships of haplochromines is only the first step in their conservation. Preserving the integrity of species distinctions in captivity may be even more difficult. There is little evidence of interspecific hybridization of haplochromines within Lake Victoria, but preliminary observations strongly indicate that hybridization is both possible and probable within the confines of aquariums less than several thousand liters in volume. Interfertility among hybrids appears to be common, and bizarre hybrids between morphologically disparate haplochromine species have been observed in hobbyists' aquariums. Similarly, entirely distinct Malawiian cichlids that have never been known to hybridize in nature, do so readily in aquariums; such "homogenization" of Malawi mbuna stock's is a common and familiar problem in large cichlid exhibits at public aquariums.

"In the case of Victorian haplochromines, the information base is wholly inadequate for the task of prioritizing captive propagation programs, and quite far from the stage where planning of enhancement or restoration of wild populations might be considered."

Insufficient Data

One of the most serious problems facing nascent fish conservation efforts is the extremely limited database on distribution and abundance of threatened species in the wild. In the case of Victorian haplochromines, the information base is wholly inadequate for the task of prioritizing captive propagation programs, and quite far from the stage where planning of enhancement or restoration of wild populations might be considered. What is known from recent expeditions by the British Museum, the Haplochromine Ecology Survey Team, and the Lake Victoria Research Team suggest that the situation in Lake Victoria is probably deteriorating very rapidly, but in at least some parts of the lake, shallow-water inshore cichlid assemblages do not appear to have been severely disrupted. The multinational collaboration recently established by the National Oceanic and Atmospheric Administration's (NOAA) Large Lakes of the World Program has renewed long-term ecological research on Lake Victoria.

Action Plan for Victorian Haplochromines

In hopes of consolidating conservation efforts by European, North American, and Kenyan institutions, the CBSG has drafted a five-year plan for the development of a formal SSP program for the entire Victorian endemic fish fauna (Aquarium Action Plan 1989). The objectives of the program are:

1) To establish exhibits and programs explaining freshwater extinctions, the importance of conserving native fishes, and the relationship between fish conservation and habitat preservation;

2) To stabilize, conserve, and propagate haplochromine stocks presently in North America and Europe for research and education;

3) To assess the status of Victorian cichlids in the wild, determine conservation priorities, and incorporate up to 36 of the estimated 300 haplochromine species into a faunal SSP program as is feasible and appropriate.

Additional key elements of the plan include:

• Networking of the major organizations to be involved in CBSG activities, including the American Association of Zoological Parks and Aquariums, the IUCN, the U.S. Fish and Wildlife Service, and corresponding agencies in other countries;

• Acquisition of all stock of known origin now in captivity; and

• Development of a 35mm slide and specimen reference collection at the Harvard Museum of Comparative Zoology.

Proposed research activities include: long-term field studies in collaboration with Kenya, Tanzania, and Uganda to assess the changing status of endemic species in the lake, and explore possibilities for long-term on-site conservation measures; a survey of the nature and scope of developmental plasticity in a variety of haplochromine species; the development of protocols for measuring genomic diversity in captive and wild stocks; and the exploration of cryopreservation of gametes as a management tool for cichlid stocks.

Conclusion

To gain perspective, one should consider multiplying the scope of the program outlined above by the number of freshwater fish faunas likely to become endangered in coming decades. The probability of maintaining intact haplochromine communities in Lake Victoria is extremely low, and the odds of restoration or effective conservation management are highly uncertain. However, the research program necessary to conserve a portion of this assemblage is of general theoretical interest, and is likely to yield important results for evolutionary, molecular and conservation biology. For this reason alone, a portion of the fauna should be maintained in the wild, and as a last result through captive propagation.

The creation of an SSC captive breeding specialist group and initiation of field laboratory research on Lake Victoria fishes are important steps. We're grateful for the support of NOAA, the National Science Foundation, the Institute for Museum Sciences, the Environmental Protection Agency, and the Pew Scholars Program for Conservation and the Environment for their generous support for our field and laboratory research and captive breeding program. This project is a collaboration of scientists from Kenya, Uganda, Tanzania, the United States, Holland, Great Britain, and Israel, and involves over 30 participating museums, universities, and public aquariums. Yet, aquariums cannot hope to serve as arks for more than a very small proportion of threatened freshwater fish diversity. In the long term, the goals of the Captive Breeding Specialist Groups must eventually turn back to the fostering of good environmental stewardship — through education, through the transfer of technologies that foster renewable exploitation of native species, and through continuing basic research. The tasks before us are daunting — their contemplation is a depressing task in itself. But we have begun.

"... aquariums cannot hope to serve as arks for more than a very small proportion of threatened freshwater fish diversity. In the long term, the goals of the Captive Breeding Specialist Groups must eventually turn back to the fostering of good environmental stewardship..."

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Note: This paper is a modified and updated version of a paper published in the proceedings of the 5th World Conference on Breeding Endangered Species in Captivity in Cincinnati, Ohio, in 1988, edited by B.L. Dresser, R.W. Reece, and E.J. Maruska (see Kaufman 1988).

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Apache Trout Culture: An Aid to Restoration

by
Bob David

The Apache trout, *Oncorhynchus apache*, formerly *Salmo apache*, historically inhabited streams in the White Mountains of east central Arizona, including the headwaters of the Salt, San Francisco, and Little Colorado Rivers. Although the species has been noted in the literature since the late 1800s, it was not until 1972 that Robert R. Miller of the University of Michigan first described and recognized the Apache trout as a separate species.

The appearance of the Apache trout is unique among most North American salmonids. Its olive-green dorsal surface blends to a brassy gold coloration on the sides, gill covers, and fins. Dorsal, pelvic, and anal fins are edged in white and occasionally display orange tips, depending on the carotenoid content of the diet. The Apache trout also displays the largest dorsal and adipose fins of any other trout (*Oncorhynchus* spp.). Spots are normally scarce and frequently outlined with a noticeable pale halo. The light yellow belly may also harbor a few spots and often exhibits blotches of orange in both sexes, particularly during the spawning season. A yellow "cutthroat" slash is also present in the gular folds, however, the ancestry of this species is linked more closely to the rainbow trout than that of the cutthroat.

Researchers generally believe that the Apache trout (also known as the Arizona trout, Arizona golden trout, Arizona native trout) once inhabited most perennial streams above 1,827 meters within its historical range. However, beginning in the late 19th century, human interactions began to have a marked effect on populations. Grazing, logging, road construction, and other habitat impacts altered watersheds, causing increased temperatures and siltation, which are detrimental to the spring-spawning Apache trout. Stocking of exotic trouts, which began as early as 1917, also contributed to the decline of the species. These highly efficient exotic competitors have included the rainbow, brown, brook, and Yellowstone cutthroat trouts, of which the rainbow and cutthroat trouts are able to hybridize with the Apache trout. This hybridization, along with competition, is attributed as a major factor in the extirpation of the Apache trout from many streams which it formerly inhabited.

Three-year old female Apache trout at Alchesay-Williams Creek National Fish Hatchery
Photo: B. David, USFWS

"The production and distribution of large numbers of Apache trout, however, is not the final answer in the recovery of this threatened species. Artificial propagation is only a single tool among many in the overall plan to delist the species."

Protection Efforts

The preservation of the Apache trout was initially undertaken by the White Mountain Apache Tribe in the late 1940s. At that time it was believed that the remaining pure populations in existence were present in only a few streams on the Fort Apache Indian Reservation. In 1955, all streams on the Reservation thought to contain pure populations were closed to fishing. During the 1960s, extensive surveys were conducted by the Arizona Game and Fish Department (AGFD) and the U.S. Fish and Wildlife Service (USFWS), which led to a better understanding of the status of the species. It was also during this time that the AGFD relocated fish from Ord Creek, a population located on the Fort Apache Indian Reservation, to its Sterling Springs State Fish Hatchery. This was the beginning of captive rearing efforts designed to assist in the efforts to recover the species.

The Apache trout was originally listed as "endangered" under the Endangered Species Act of 1973. However, following the recommendation of the Recovery Team, the listing was changed to "threatened" in order to assist management and culture efforts. In addition, the Recovery Team also set a goal for the restoration of 30 discrete populations of the species within its historic range, as a condition for delisting.

Captive Breeding Efforts

Attempts at culturing Apache trout by the State of Arizona in the early 1960s and 1970s were largely unsuccessful due to the conditions and methods applied. Culture techniques currently in use on domestic rainbow trout strains were tried on wild Apache trout with very poor results. High mortalities and the inability of the species to utilize commercial trout diets eventually led to the disbanding of the program in 1981.

Initial research on controlled, artificial propagation of Apache trout by the USFWS began at the Alchesay-Williams Creek National Fish Hatchery Complex in May of 1983. Wild Apache trout were spawned on-site from the East Fork of the White River. Spawning personnel were airlifted into the remote site, and fertilized eggs delivered to the Williams Creek Station by helicopter. Eggs were hatched, and fry were subjected to trials designed to determine suitability of introductory feeds. Out of 2,715 eggs collected, only 240 hatched and survived to be used as future broodstock. During May 1984, wild Apache trout were again spawned on site from the East Fork of the White River. Out of 1,869 eggs collected 1,204 were hatched, and 704 two-inch fish were able to be retained as broodstock. These two-year classes of wild fish formed the basis for all future culture work at the Williams Creek Station.

Reintroduction Program

Herein began a rather ambitious goal to replace all rainbow trout stocking on the Fort Apache Indian Reservation with similar programs using only the Apache trout. The rainbow trout program had involved the stocking of approximately 500,000 catchable (20 cm), subcatchable (15 cm), and fingerling (8 cm) rainbow trout annually in waters consisting of over 1,200 hectares of lakes and 650 kilometers of streams. This goal is not entirely a recovery effort, but a program designed to enhance current sport fishing enterprises while complimenting recovery plans. In addition to providing for a sport fishery, the program also supplies fingerling Apache trout to the AGFD for their use in restoring a portion of the targeted 30 streams prior to delisting of the species.

State-of-the-art techniques were used in developing culture methods for Apache trout. These included the use of oxygen injection into the female body cavity to expel eggs (air spawning). Milt was collected from anesthetized males by aspiration, subjected to short-term storage, and classified as to viability prior to fertilization. Resulting fry were fed in trials using the latest high-performance diets designed for larval development of warm water species found difficult to culture, such as walleye and muskellunge. Trials resulted in the selection of a specific semi-moist diet, introduced to the fry by use of remote, mechanical feeders controlled by programmed timers.

The use of these, and other culture techniques led to the initial spawning and culture of 1,200 Apache trout in 1986, 22,000 trout in 1987, 90,000 in 1988, 175,000 in 1989, and 550,000 in 1990. Apache trout production in 1990 will result in the accomplishment of the initial goal of replacing all rainbow trout stocking with Apache trout by 1991.

Evaluation of Captive Breeding and Reintroduction Strategy

The production and distribution of large numbers of Apache trout, however, is not the final answer in the recovery of this threatened species. Artificial propagation is only a single tool among many in the overall plan to delist the species. Stocking of a native species in historical waters does nothing to repair damaged habitat, nor will its eggs be able to survive the onslaught of silt common to present spring run-offs. Stocking must be preceded by habitat improvement projects designed to restore watersheds to a condition conducive to reproduction and survival of native species.

"Stocking of a native species in historical waters does nothing to repair damaged habitat, nor will its eggs be able to survive the onslaught of silt common to present spring run-offs. Stocking must be preceded by habitat improvement projects . . ."

In addition, we must be concerned with the genetic integrity of the populations we choose to propagate. Artificial culture should not impair the ability of a species to re-establish itself in a wild state. This can be ensured through the use of biochemical systematics to monitor the continuity of a polymorphic gene pool. Unfortunately, pure populations of Apache trout remain only in isolated headwater regions, cut off from other populations, where they have evolved into relatively homozygous entities. There is some question now concerning their ability to adapt to other waters with conditions different from that in which they currently exist. This is another reason why it is important to preserve what little polymorphism remains in the species by attempting to avoid genetic loss due to selection during artificial propagation. One strategy to ensure this integrity is the periodic introduction of genetic material from the wild into captive populations. In the case of Apache trout, alteration of spawning times between wild and captive populations has made this difficult. It is hoped that cryopreservation of gametes from wild stock may be used to assist in this process.

Currently, research is being conducted on the survival of reintroduced Apache trout. Marked fish have been placed in both lake and stream environments in order to determine their ability to adapt to different environmental conditions and their ability to compete with other species. Only preliminary data have been gathered, however, there are some indications that winter survival by

Apache trout in stream environments may exceed that exhibited by exotic species such as the rainbow trout. Survival in lake environments may favor introduced species. The temperature, oxygen, and chemical changes inherent in lakes favor a species with a broad, polymorphic gene pool. Thus, Apache trout, which evolved in conditions of high oxygen concentrations and relatively stable chemical balances, may not be able to compete in these environments.

Restoration of streams through re-establishment of historical species composition is not always a popular management objective. In many cases, sportsman groups are opposed to the loss of existing fisheries composed of exotic species which are providing a significant sport fishery. Past experiences have seen these sites used in endangered species restoration, only to be closed to sport fishing for extended periods while small reintroduced numbers of fish expand their populations to a size capable of supporting a fishery. Artificial propagation may be used to shorten the time a stream remains unavailable to fishermen by stocking numbers approaching carrying capacity. In this way we are able to gain public support and possibly proceed in restoring a site that may not otherwise be considered due to adverse public opinion.

"Restoration of streams through re-establishment of historical species composition is not always a popular management objective. In many cases, sportsman groups are opposed to the loss of existing fisheries composed of exotic species . . ."

Another area in which artificial propagation may prove to be a valuable tool is in the proposed technique of gene swamping. This involves the stocking of large numbers of endangered species on top of existing exotic populations. While hybrid populations may result, it is hypothesized that natural attrition, through several generations, may produce a relatively pure population. This application would be restricted to restoration of non-critical populations.

Other uses of artificial propagation in the recovery of endangered trouts have yet to be defined. Biologists should be encouraged to explore further possibilities for the use of this management tool. Through careful management of individual gene pools, culture of endangered or threatened species can greatly enhance a recovery effort.

"Artificial propagation may be used to shorten the time a stream remains unavailable to fishermen by stocking numbers approaching carrying capacity. In this way we are able to gain public support . . ."

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The Red Wolf: Recovery of an Endangered Species

by
Michael K. Phillips

Species Decline

Red wolves (*Canis rufus*) ranged throughout the southeastern United States before European settlement of that region. However, by 1980, the species was considered extinct in the wild (McCarley and Carley 1979). Demise of the red wolf was due to many factors: human persecution of wild canids and destruction of habitat forced the last few red wolves to use marginal habitat in Louisiana and Texas where they bred with coyotes (*Canis latrans*) and suffered heavy parasite infestation (Carley 1975). Although the plight of the red wolf was recognized in the early 1960s (McCarley 1962), the species was not listed as endangered until 1967, and did not receive "priority treatment" until passage of the Endangered Species Act in 1973. In that year the U.S. Fish and Wildlife Service (USFWS) initiated a recovery program.

Red wolf from Alligator River Photo: USFWS

"By the mid-1970s, the USFWS realized it was not possible to preserve the species in the wild, and concluded that recovery could only be achieved through captive breeding and reintroductions."

Captive Breeding and Reintroducation Efforts

By the mid-1970s, the USFWS realized it was not possible to preserve the species in the wild, and concluded that recovery could only be achieved through captive breeding and reintroductions. By the late 1970s, the captive breeding program was established utilizing 17 pure red wolves that were captured in the Texas and Louisiana refugium (USFWS in press). Initially the program languished due to a limited federal budget (about $30,000 per year) and little interest from the zoo community. The Point Defiance Zoo and Aquarium in Tacoma, Washington, initially was the only zoological facility involved. If not for the commitment of a handful of people and the Point Defiance Zoo, the red wolf would have slipped into oblivion.

By 1985, six zoological facilities held red wolves and the captive population had grown to 65 individuals. With the species secure, the USFWS intensified recovery efforts by initiating a reintroduction project in northeastern North Carolina. Because the reintroduction represented the first attempt in history to restore a carnivore species that was determined to be extinct in its former range (McCarley and Carley 1979), the project generated tremendous interest about the prospects of recovering the species. From the fall of 1987 through October 1990, 14 more zoological facilities committed themselves to maintaining red wolves, and the annual federal budget for captive breeding increased to about $200,000. As of October 1, 1990, 131 red wolves existed in captivity. The revised Red Wolf Recovery Plan (U.S. Fish and Wildlife Service in press) calls for increasing the population to 330 animals.

The reintroduction project in northeastern North Carolina occurred within the confines of the 63,636 ha Alligator River National Wildlife Refuge (ARNWR) and adjacent Department of Defense land covering an additional 20,454 ha. The areas consist of marshes, nonriverine swamp forests, pocosins, and agricultural fields. Local climate is characterized by hot summers, mild winters, and high humidity. The areas are bisected by numerous logging roads in various stages of development/repair; many are only seasonally passable with a 4-wheel drive vehicle or on foot.

Before release, each wolf was acclimated to the refuge. The length of acclimation varied from a few weeks to 2+ years. During acclimation we minimized human contact, hoping to reduce the wolves' tolerance of humans, varied the feeding regime to expose the animals to feast or famine, weaned the wolves from dog food and fed them an all meat diet, and provided the opportunity to hone predatory skills by giving them live prey.

Before the wolves were released, each was given a health check and fitted with a motion sensitive radio-collar. Nine of the first ten wolves released were implanted with radioactive tags (Crabtree et al. 1989). The tags allowed us to assign collected scats to individual wolves. Additionally, the first nine wolves were implanted with abdominal transmitters. These transmitters were placed as backups to the radio-collars. Pups which were too small to wear a radio-collar at the time of release were also implanted with abdominal transmitters.

Species Response

Between September 1987, to mid-October 1990, 29 captive-born wolves (19 adults and 10 pups) were released on 13 occasions (Smith and Phillips 1987, Phillips 1988, Phillips and Parker 1988). In addition, a minimum of six pups born in the wild during 1988 and 1990 were monitored by USFWS personnel (Phillips 1989). Wild-born offspring are irrefutable evidence that captive-born-and-reared adults can make the transition from captivity to life in the wild. As of mid-October 1990, a minimum of 19 wolves were free-ranging in northeastern North Carolina.

After varying lengths of time, most released wolves settled into home ranges that varied in size from 50 km^2 to 100 km^2; small home ranges were situated in agricultural areas, whereas large home ranges were situated in forested regions. Mated pairs actively defined home ranges through scent-marking and howling. These behaviors, along with intraspecific strife, suggest that reintroduced red wolves will occupy exclusive home ranges. As a result, sociality is expected to significantly affect the size of the reintroduced population.

Reintroduced wolves were crepuscular and more active at night than during the day. Analysis of 1,100 scats indicated that white-tailed deer (*Odocoileus virginianus*), raccoon (*Procyon lotor*), and marsh rabbits (*Sylvilagus palustris*) were important food items. Because wolves were released in areas containing no or few coyotes, biologists are uncertain how these two species may interact in the future.

During the first three years of the reintroduction effort, one animal was returned to captivity and not re-released, and 15 wolves died: five animals were killed by vehicles, two by other wolves, one of a uterine infection, one of pleural effusion and internal bleeding, one suffocated after a raccoon kidney became

USFWS personnel handling a 57-day old red wolf pup Photo: USFWS

lodged in his trachea, one drowned after being captured in a leghold trap set by a fur trapper, and four drowned after apparently trying to cross a wide expanse of water north of ARNWR. The USFWS believes it is a measure of the program's success that all the deaths were natural or accidental, and apparently not the result of a citizen acting irresponsibly, motivated by some unfound hatred for wolves.

"... in part to enlist support from local sportsmen, the USFWS decided to permit hunting and trapping of other game species in the reintroduction area, even though such activities may result in the accidental 'take' of a red wolf."

Clearly, vehicles are an important source of mortality. To alert motorists to the presence of wolves along highways, the North Carolina Department of transportation erected red wolf crossing signs and local radio stations began to air public service announcements.

The reintroduction of red wolves would not have been possible without public support, which was cultivated in part through considerable pre-release consultation between the USFWS and interested groups. The USFWS briefed representatives of environmental organizations in Washington, DC, the North Carolina Congressional Delegation, the North Carolina Department of Agriculture, the Governor's office, local officials, and local landowners. The U.S. Air Force and Navy were briefed because they conduct training missions on 17,716 ha adjacent to the refuge. Numerous personal contacts were made with local citizens, especially hunters and trappers, in preparation for four public meetings held during February 1986.

Experimental Population Status

At the briefings/meetings, considerable effort was spent explaining the significance of the decision to consider reintroduced wolves as members of a "experimental/nonessential" population (Parker and Phillips in press). The experimental/nonessential designation was promulgated under the 1982 amendments to the Endangered Species Act (Public Law No. 97-304), and provided the USFWS with the ability to relax restrictions of the Act to encourage cooperation with reintroduction projects from those likely to be affected by them. For example, in part to enlist support from local sportsmen, the USFWS decided to permit hunting and trapping of other game species in the reintroduction area, even though such activities may result in the accidental "take" of a red wolf. The USFWS decided that prosecution would not be pursued when taking of a red wolf was unavoidable, unintentional, or did not result from negligent conduct — provided that the incident was reported immediately to the refuge manager (or other authorized personnel). The USFWS further decided that wolves could be taken in defense of human life, but not to prevent or reduce depredations (e.g., of livestock, chicken, or pets). In instances of depredations, citizens are required to contact USFWS or state conservation officers authorized to institute control measures. Fortunately, to date no depredations have occurred.

Program Success

Not surprisingly, the reintroduction project attracted considerable interest from the media and private citizens (Phillips 1990). Since the fall of 1986, a minimum of 22 magazines and 24 newspapers published stories about the project. In addition, the project was discussed during the nightly newscasts of five national and four regional television networks, and was the focus of four television documentaries.

Thirty-three private citizens donated approximately 10,000 hours of volunteer time to the project. Local civic groups helped with fund-raising. Three landowners entered into agreements with the USFWS that provided wolves access to an additional 20,000 ha adjacent to the refuge. The restoration project now covers approximately 106,000 ha of federal and private land.

Success at ARNWR spawned several ancillary projects. For example, red wolves have been released on three islands off the southeast coast. From 1988 through October 1990, ten animals were released on Bulls Island, a component of the Cape Romain National Wildlife Refuge off the coast of South Carolina; currently two animals inhabit the island. During July 1989, nine wolves were released on Horn Island, a component of the Gulf Island National Seashores off the coast of Mississippi; currently five animals inhabit the island. During August 1990, four wolves were released on the St. Vincent National Wildlife Refuge, an island off the northwest coast of Florida; all four are still free-ranging. The island projects' primary objective is to provide young wild-reared wolves for mainland reintroduction sites. Currently the USFWS is planning to initiate a second mainland project in the Smoky Mountains National Park during August 1991.

"... insufficient habitat and habitat destruction per se are not the limiting factors for red wolf recovery . . . The problem is that much of the available suitable habitat in the southeast U.S. is privately owned. Landowner support is therefore required. . ."

Because red wolves are "generalists" (e.g., in their diet requirements), they can flourish in a wide variety of habitats. Hence, insufficient habitat and habitat destruction per se are not the limiting factors for red wolf recovery; there is plenty of habitat available to meet the population recovery objectives as outlined in the USFWS recovery plan. The problem is that much of the available suitable habitat in the southeast U.S. is privately owned. Landowner support is therefore required for successful reintroduction initiatives. Thus, unlike most endangered species, recovery of the red wolf is not so much dependent on the setting aside of undisturbed habitat as it is on overcoming the political and logistical obstacles to human coexistence with wild wolves.

The revised Red Wolf Recovery Plan (U.S. Fish and Wildlife Service in press) calls for maintaining 330 animals in captivity and 220 animals in the wild at a minimum of three different mainland locations. Because of captive breeding and subsequent reintroductions, the red wolf recovery project enjoys tremendous local, regional, and national support. USFWS personnel working on the project are confident that current momentum will thrust the species to the edge of recovery within 10 to 20 years.

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The Conservation Program for the Golden Lion Tamarin, *Leontopithecus rosalia*

by
Devra G. Kleiman, Benjamin B. Beck, Andrew J. Baker, Jonathan D. Ballou, Lou Ann Dietz, and James M. Dietz

The Golden Lion Tamarin Conservation Program's (GLTCP) ultimate goal is the survival of the golden lion tamarin and the preservation of Atlantic Coastal Rainforest habitat. This is a unique multidisciplinary effort, involving pure research and applied conservation. The

Golden lion tamarin   Photo: Jessie Cohen, NZP

separate components of the program include:

1) studies of the demography, population dynamics, behavioral ecology, genetics, physiology, nutrition, reproduction and social behavior of golden lion tamarins in the wild;

2) biological studies of captive golden lion tamarins and the management of the captive population to preserve genetic variability;

3) the development of a public education program involving the local and national community to gain support for the conservation of species and natural habitats, and the training of Brazilian nationals in the area of conservation biology and wildlife management to meet future needs in Brazil;

4) the protection, management, and expansion of the Poco das Antas Biological Reserve and surrounding habitat areas, and research on restoration techniques for rehabilitatation and reforestation of degraded tropical habitats;

5) the development and refinement of techniques for reintroducing captive-born golden lion tamarins into the wild, and translocating wildborn tamarins into protected habitats; and

6) biological studies of species sympatric with golden lion tamarins in order to understand better the community ecology of the Atlantic Coastal Rainforest, to evaluate the condition of available habitats for tamarin translocations and reintroductions, and to increase our understanding of the life histories of other rare, endangered or endemic fauna of this region.

The Wild Population

Historically, golden lion tamarins were distributed along the lowland coastal forest of Rio de Janeiro State, possibly extending into southern Espirito Santo. Deforestation in this area for agriculture, timber, and charcoal has reduced available golden lion tamarin habitat to isolated forest patches in five municipalities within Rio de Janeiro State. Few of these patches exceed 1,000 hectares in size. The only officially protected area containing golden lion tamarins is the Poco das Antas Biological Reserve in the municipality of Silva Jardim. The Reserve is approximately 5,300 hectares in area, but because of deforestation prior to the creation of the Reserve, only an estimated 3,000 hectares currently offers habitat suitable for tamarins. Although there have been no known cases of poaching of wild tamarins in the Reserve since 1983, tamarins are still being poached and are entering the illegal animal trade in Brazil. Deforestation, however, has been reduced significantly in recent years.

Because it was created from several cattle ranches, the Poco das Antas Reserve is only about 40% forested. Forest regeneration was retarded due to annual fires during the dry season. However, the construction of firebreaks in the mid-1980s exerted some control over the spread of fires until February 1990, when an especially dry wet season, together with strong winds, resulted in a severe fire that destroyed regenerating vegetation in an area covering over 25% of the Reserve. Studies by biologists from the University of Georgia are documenting forest regeneration and the effects of the fire.

The GLTCP also supports studies of other species endemic to the Atlantic Coastal Rainforest of Brazil, partly to increase our knowledge of some of these rare and endangered forms, but also to understand better the animal and plant community in which golden lion tamarins live. Colleagues have conducted projects on the endangered maned sloth (*Bradypus torquatus*) and the little-known bamboo rat (*Kannabateomys amblyonyx*).

The current population estimate of golden lion tamarins in the Poco das Antas Reserve is 290. This estimate has risen steadily since the first estimate of 75 in the late 1970s. This trend probably reflects both better information and a real population increase over the past decade. The current estimate of the population outside the Reserve, excluding reintroduced animals, is 550, distributed among two government-owned areas of 800 and 1,500 hectares, and a number of privately-owned areas.

Studies initiated in 1983 on the basic biology of *Leontopithecus rosalia* have resulted in extensive information on habitat requirements, territory size, and social structure. This information has been critical to identification of potential habitat (e.g., for reintroduction and translocation), estimates of current populations, formulation of survey techniques, and planning strategies for integration of native and reintroduced populations. Also, through these studies, we have established that the Poco das Antas population is demographically healthy, with a high birth rate and an infant survival rate higher than that seen in the captive population. However, the Reserve is not large enough to maintain a genetically viable population of tamarins over the next 50 to 100 years without inbreeding problems. Currently, the Reserve population is probably at or near the carrying capacity of the habitat.

History of the Captive Population

The collaborative management of the captive population of golden lion tamarins began in 1972, when the Wild Animal Propagation Trust convened an an international conference to evaluate the status of the golden lion tamarin in captivity and the wild. At that time, there were fewer than 80 individuals in captivity, and colonies at different institutions were managed independently of each other. Mortality and natality rates indicated that the population was not self-sustaining.

Following the conference, staff from the National Zoo initiated an intensive research program on captive individuals to determine the cause of the mortality and reproductive problems plaguing the population. At the same time, an International Studbook was developed by Marvin Jones to track information on the pedigree and life-history data for each individual in the population.

Between 1972 and 1975, the population grew slowly, but demographic analyses suggested that it was still headed for extinction. However, research focusing on the behavior, nutrition and husbandry of the species finally began to have an impact. Major advances were made when findings revealed that reproduction is most successful when animals are maintained in monogamous pairs and offspring are kept in the family groups to assist parents in rearing younger siblings. In addition, the high protein requirements of tamarins became clear, resulting in a major diet change for the zoo population.

With a better understanding of the husbandry requirements of the species, reproduction increased. In 1981, an International Management Committee was formed to coordinate the global management of the captive population and assure that it was managed according to sound genetic and demographic guidelines. Strict husbandry and management protocols were developed for institutions holding golden lion tamarins. Any institution interested in acquiring the species was required to apply to the Committee for approval, and sign a Cooperative Research and Management Agreement. An essential element of the Agreement was that no tamarins were to be used in commercial transactions. In the same year, the golden lion tamarin was included as one of the first species in the American Association of Zoological Parks and Aquariums' Species Survival Plan.

Some of the authors with a tamarin Photo: J. Cohen, NZP

By 1983, the captive population had grown to over 370 individuals. Efforts were begun to slow the population growth (which was as high as 25% per year) by reducing the number of breeding pairs, and controlling the number of young produced by each pair through the use of contraceptive implants.

Today, the captive population of golden lion tamarins is the most intensively managed global population of any species in captivity. The population consists of over 560 animals distributed in 119 institutions worldwide. Population management, based on demographic and genetic analyses of the population, determine who is to be paired with whom, how many offspring they are to produce and when.

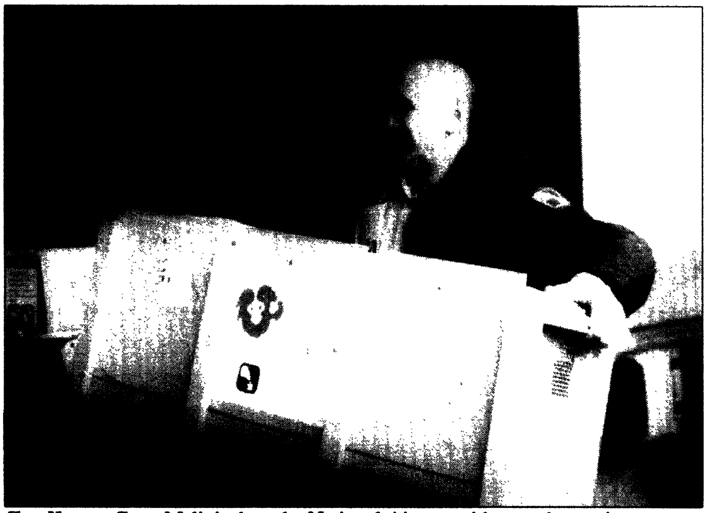
"Today the captive population of golden lion tamarins is the most intensively managed global population of any species in captivity."

The Reintroduction

Once the breeding of the captive population had achieved some degree of regularity and control, discussions were initiated in 1982 with the Brazilian government (IBDF), the Rio de Janeiro Primate Center (CPRJ-FEEMA), and other interested parties concerning the possibility of starting a reintroduction program using captive-born animals that were surplus to the needs of the captive population. These institutions and their staffs have worked and continue to collaborate closely with us.

Between 1984 and September 1990, we have reintroduced 75 zoo-born and six wild-born golden lion tamarins. The wild-born tamarins had been captured illegally as pets and subsequently confiscated by Brazilian authorities. Twenty-

"Population management, based on demographic and genetic analyses of the population, determine who is to be paired with whom, how many offspring they are to produce and when."



Zoo Keeper Gene Maliniack at the National Airport with crated tamarins
Photo: Jessie Cohen, Natiobal Zoological Park, Smithsonian Institution

seven of the zoo-borns and four of the wild-borns still survive. There have been 39 births in the wild to reintroduced parents; 26 infants survive. Reintroduced tamarins and their offspring live in 15 groups with reproductive potential in the Poco das Antas Biological Reserve and on seven adjacent privately owned ranches. Some live totally without provisioning; others are visited and fed by observers daily. Losses of reintroduced tamarins have resulted from starvation, exposure, disease, predation, bee sting, snakebite, wounding in intraspecific aggression, and theft by humans. Most losses were due ultimately to lack of recognition of natural foods, lack of recognition of non-avian predators and dangerous non-predatory animals, deficient locomotion, and poor spatial orientation.

Pre-release training in such skills, which was very labor-intensive, conferred only a transient advantage and was deemed not cost-effective. Instead, tamarins are now released shortly after shipment to Brazil with little training or acclimation. They are provisioned and otherwise supported intensively for 18 months as they learn to forage, locomote and orient in the forest itself.

Survival is inversely correlated with age. Intact family groups with two to four subadults survive longer than adult pairs. Pairing a previously reintroduced subadult with an opposite-sexed new subadult reintroductee increases survival and accelerates natural foraging and locomotion.

Public and Professional Education

One of the major goals of this program is the training of a cadre of professional Brazilian conservation biologists. We have been recruiting students to the GLTCP on a regular basis and have made our facilities available for several wildlife management and conservation biology training programs. We also are collaborating with the Universidade Federal de Minas Gerais in Belo Horizonte.

In addition, we have, as an important part of our activities, a community education program. Without the collaboration of the people in the region where golden lion tamarins still exist, all other efforts will have limited results for the long-term conservation of the species. This community education program is developing a model for the solution of conservation problems locally, nationally in Brazil, and internationally.

The education program is best understood in terms of a systems model; the model helps to focus efforts on priority problems, systematically develop solutions, and ensure the continued effectiveness of the solutions. We first defined the priorities by addressing local conservation problems. Specific objectives which we set include: reducing deforestation, assuring the permanent conservation of privately-owned remnant forests in the region, reducing fire in the Poco das Antas Reserve, reducing the illegal commerce of golden lion tamarins, and reducing illegal hunting in the Reserve.

We then identified and evaluated the population, resources, and setting for our activities -- a process somewhat like market research. To understand the population better, we conducted interviews about local knowledge, and attitudes and behavior towards the forest, local wildlife, and the Reserve. We also determined that the best methods for reaching the local community were through the mass media, i.e., television and radio. To build a positive relationship with the community, we encouraged community involvement in the planning and implementation of project activities, and began developing interactions with all sectors of the community, including teachers, politicians, private land owners, businessmen, etc.

We then selected educational materials and activities which interested the local leaders and which seemed most likely to have the widest results for the least cost. We produced printed material for schools, did audio-visual and live presentations, and developed a variety of materials after first testing prototypes.

We worked with a single community initially to test the effectiveness of our programs, and then expanded our reach. We have, to date, conducted activities with schools, farmworkers, Reserve guards, and the press, to name a few specific groups, as well as the entire community through larger events such as parades.

The education program sought to evaluate the effectiveness of the separate components of the program by a second set of questionnaires after two years of community activities. Results suggested that there were significant changes in the attitudes and knowledge of local Brazilian adults. For example, to support a species, a public needs to be able to identify it. There was a significant increase in the percentage of people who recognized and could identify a photograph of a golden lion tamarin two years after the initiation of our educational program. We are now in the stage of altering our programs in keeping with our survey results to improve the cost-effectiveness of our activities, expanding our activities to other sites, and training individuals who can begin to develop comparable programs in other regions of Brazil.

Conclusion

In the case of the GLTCP, we feel that the combination of activities and the interaction among the separate components of the program have resulted in much greater success towards our ultimate goal of preserving golden lion tamarins and the Atlantic Coastal Rainforest than if we had concentrated on only one or two aspects. We also believe that this multidisciplinary effort is a model worth following for any organizations intending to pursue a captive breeding/reintroduction effort in future.

Acknowledgements

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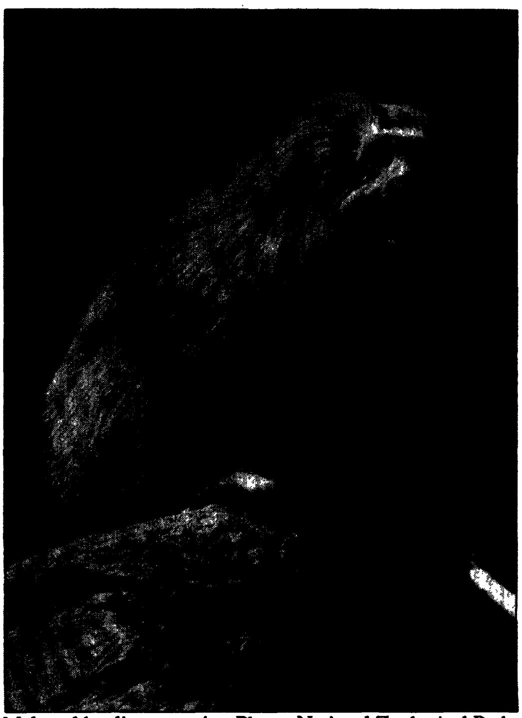

Male golden lion tamarin Photo: National Zoological Park

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Black-Footed Ferrets On The Road To Recovery

by

Tim W. Clark

Introduction

The black-footed ferret restoration effort has been long, problematic, and at times tenuous. However, currently the program is on the right road for full species recovery. This paper outlines some historical milestones of the recent ferret recovery effort. Much of the progress is summarized in an upcoming annotated bibliography on the black-footed ferret which lists 118 articles published between 1986 and 1990 (Reading and Clark in press).

The black-footed ferret (*Mustela nigripes*) is a medium-sized mustelid of the Great Plains that appears to be an obligatory predator on prairie dogs (*Cynomys* spp.) (Clark 1989). Because of their presumed competition with agriculture and livestock, these rodents have been exterminated by widespread poisoning, shooting, and other means for nearly a century and have been reduced by 70 to 90+% in both numbers and area over their entire original range. These legally mandated extermination programs pose a continuing limitation on planned ferret reintroductions.

About 1980, the ferret was considered extinct by many people. But discovery of a small population near Meeteetse, Wyoming, in 1981 brought hope for species recovery. Nearly 300 ferrets were studied over the next few years, with a peak population of 129 in 1984. Catastrophe struck in 1985 when naturally contracted canine distemper killed most ferrets, leading to a species low of about ten individuals in 1986.

The ferret is extinct in the wild today and only exists in three captive populations totalling about 181 animals. Continued increases in captive populations, reintroduction, and successful management — including restoration and long-term management of the prairie dog ecosystem (Clark et al. 1989, Miller et al. in press) — offer the only road for restoration.

History of Recent Restoration Efforts

Discussion about how to restore ferret populations was part of the Meeteetse ferret studies from the beginning. In 1983, Richardson et al. (1986) first formally examined recovery options and strongly recommended the captive-rearing/translocation option, and called for an interagency meeting soon thereafter to decide on a strategy. In 1984, Carpenter (1985:12.1) summarized efforts in the previous decade to captive breed ferrets and concluded that "it would appear that a successful captive breeding program . . . can be realized as soon as a founder population . . . can be taken into captivity." In 1984, Clark et al. (1985:7.11) noted that the "single [Meeteetse] ferret population is currently highly vulnerable to extinction" and that "captive breeding/translocation efforts . . . should begin this fall." Later, Forrest et al. (1985) outlined ferret management and reintroduction considerations. In May 1985, state and federal agency administrators decided to take a few ferrets into captivity later in the year if the wild population was large enough (Thorne and Oakleaf in press). Unfortunately, by the time six ferrets were captured, nearly all of the wild population had died of canine distemper. After this tragic start to captive breeding, another six ferrets were successfully taken into captivity; between then and February 1987, when the last wild ferret was captured, a total of 18 wild ferrets were taken into captivity. Today's ferret population comes from these few wild founders. The fate of the species now rests on captive breeding and restoration.

Adult male black-footed ferret — Photo by USFWS, Denver Wildlife Research Center

Plans for Ferret Restoration

The ferret is federally listed under the Endangered Species Act, thus the U.S. Fish and Wildlife Service (FWS) is required to ensure its survival. In early 1982, the FWS requested that the Wyoming Game and Fish Department (WGFD) assume "lead agency" status for ferret recovery in Wyoming. As a result, the state controlled the pace and direction of the national recovery effort. By 1987, Wyoming produced a Strategic Plan, the 1991/1992 objectives of which included: "Initiate experimental reintroduction of black-footed ferrets by 1991 and a full scale reintroduction program of more than 50 black-footed ferrets per year by 1992" (WGFD 1987:5). The 1996 objective was to: "Reestablish at least 2 black-footed ferret wild populations in Wyoming and maintain these populations with a total of at least 80 breeding-aged adults" (WGFD 1987:5).

The 1988 FWS Recovery Plan objective is: "To ensure immediate survival of the black-footed ferret by: 1) Increasing the captive population of black-footed ferrets to a census size of 200 breeding adults by 1991; 2) Establishing a prebreeding census population of 1,500 free-ranging black-footed ferret breeding adults in 10 or more populations with no fewer than 30 breeding adults in any population by the year 2010; and 3) Encouraging the widest possible distribution of reintroduced black-footed ferret populations" (FWS 1988:19). Ferret increases in captivity and plans for reintroduction are close to these schedules.

Captive Breeding Efforts

Captive breeding efforts got off to a frightening start, but have since been very encouraging. In 1985, the first six ferrets taken into captivity at a WGFD facility died of canine distemper contracted in the wild, in part because all animals were held in close proximity to one another. Unfortunately, two of the ferrets carried fatal distemper and transmitted it to the remaining four. The second six ferrets captured in 1985 were individually quarantined and survived. The 1986 breeding period produced no young ferrets. The last 12 ferrets in the wild were added to captivity in 1986 and early 1987. In 1987, two litters with seven surviving young increased the captive population to 25. In 1988, 13 litters and 34 surviving young were produced. That year the population was subdivided, with eight young going to the Henry Doorly Zoo in Omaha, Nebraska, and seven young to the National Zoological Park's Conservation and Research Center in Front Royal, Virginia. In 1989, 25 litters and 62 surviving young were produced. Additional ferrets were sent to these two zoos from Wyoming. At the time of this writing, about 70 young have been born and more are possible. In late 1990, additional ferrets may be sent to several more zoos in Kentucky, Colorado, Arizona, and Canada. Future increases in captive populations are expected.

Black-footed ferret Photo: Ray Punnovich, WCI

Much of the success of the captive breeding effort is due to advice and assistance provided by the Captive Breeding Specialist Group (CBSG) of the International Union for the Conservation of Nature and Natural Resources' Species Survival Commission. In late 1985, CBSG was invited by WGFD and FWS to aid in ferret recovery. "Members of CBSG provided expertise in genetics, management of small populations, husbandry, reproductive biology and small population biology" (Thorne and Oakleaf in press:14). In addition, CBSG assisted Wyoming and FWS "in preparation of Wyoming's strategic plan and the revised recovery plan" and in many other ways (Thorne and Oakleaf in press:14). In 1988, Wyoming and FWS terminated their formal relationship with CBSG, feeling that their mission had been met. Numerous aspects of ferret biology, physiology, and captive rearing are described in Seal et al. (1989).

Reintroduction

Reintroducing the Meeteetse ferrets to several new sites was part of the first conservation and research plans for the newly discovered ferrets (Clark 1981), and was formally recommended in 1983 and 1985, with the reports of Richardson et al. (1986) and Forrest et al. (1985). Reintroduction became formalized as the basic recovery strategy in government plans in 1987 and 1988 (WGFD 1987, FWS 1988).

As the basis for reintroductions, the Meeteetse ferret environment was described, a habitat suitability model was devised (Houston et al. 1986) and later revised (Minta and Clark 1989), and management guidelines were produced (Clark 1986). Habitat models were first applied to potential translocation sites in Montana (Clark et al. 1987). Since then, the FWS has developed another habitat model (Biggins et al. 1989), and applied it to several sites under the auspices of the Black-Footed Ferret Interstate Coordinating Committee established in 1986. The Committee meets at least once yearly and otherwise communicates through a FWS Secretariat.

Several reintroduction sites have been located throughout the western United States. One of the biggest is in northcentral Montana, estimated to be capable of supporting about 500 ferret families. Restoring ferrets to the wild is now a matter of developing successful reintroduction techniques and subsequent management procedures.

Reintroduction considerations have been the subject of much discussion and several meetings over the past few years. For example, Miller (1990) described studies using closely related Siberian ferrets as models to teach black-footed ferrets to avoid ground and aerial predators and to capture prey. Beck and Miller (in press) reviewed some implications for ferret recovery from experience with the golden lion tamarin reintroduction effort. To develop a successful reintroduction paradigm, Reading (1990) is studying a host of variables — scientific/ technical considerations, authority relations among those involved, organizational aspects, and normative features — all part of reintroduction efforts. His work is directed at providing managers with a readily accessible framework for species' reintroductions, both expediting the reintroductions and rendering the process more effective.

Mustela nigripes wearing radio collar Photo: F. Camenzind, FWS

Many details remain to be worked out before ferrets can be restored. Adequate acclimatization of ferrets to release sites will be employed. Many individual ferrets will be released at each reintroduction site. How much pre- and post-release training is needed for reintroduced ferrets is an open question. Considerable assessment and observation of released animals is imperative. These and other details need attention.

Since the research and captive breeding attempts of South Dakota ferrets in the 1960s and '70s, it has been widely acknowledged by the scientific and conservation community that captive breeding and reintroduction were necessary components of ferret recovery. With this background and the early assessment of the Wyoming ferret population status and conservation needs, Richardson et al. in 1983 (1986), Clark et al. in 1984 (1985), and Forrest et al. (1985) adamantly called for captive breeding to be initiated early in the restoration effort. It seemed clear then and now that captive breeding and reintroduction were the only strategies open for species restoration. One lesson to be learned from the ferret experience is that government officials controlling the pace and direction of endangered species programs need to understand that captive breeding is a legitimate conservation tool and needs to be used in a timely manner. A second lesson is that both captive breeding and reintroduction can be technically difficult, thus pertinent research and expertise need to be brought in early on in the process.

The Future

Successful reintroduction and management of the black-footed ferret is an opportunity to restore a unique ecosystem (Miller in press, Clark et al. 1989). Beginning in 1991 with the first reintroductions, and extending over the next decade or so, ferrets will be re-established to the prairies and intermountain basins of North America. Participants and onlookers alike hope ferret restoration will go smoothly and quickly.

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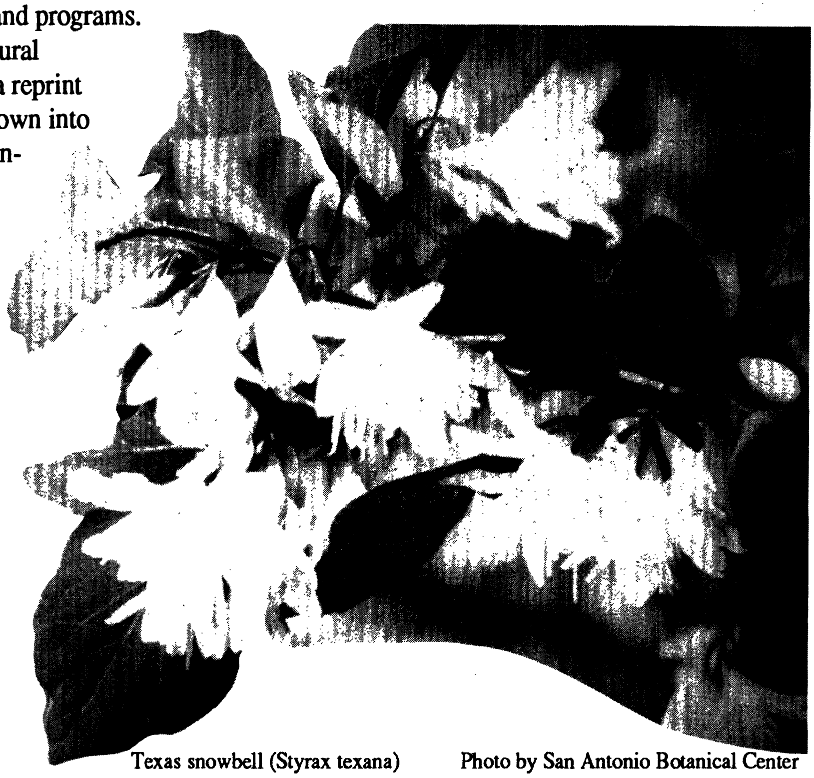

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