

Progress in Translocation of Diurnal Raptors

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ABSTRACT

Since my review in 1986, no fewer than 52 projects involving translocation (reintroduction *sensu lato*) have been conducted on at least 25 species of diurnal raptors. Using a questionnaire and a survey of literature, I evaluated these translocations and identified principal factors involved in their outcomes. Of 28 projects carried out on sufficient scale to judge degree of success, 21 resulted in establishment of breeding populations that appear viable. Most of the others show initial success—establishment of individual animals and some breeding pairs in nature. Hacking has been the main method of release, but other methods are useful. Most releases have used captive-bred young, but wild-taken young and mature birds also work. Successful translocation seems related to intensity of effort, number of birds released, and habitat quality. The higher success rate (c.75 %) in translocating raptors compared to other birds (<67%) may be explained by (1) raptors being less affected by predation, (2) physically stronger, (3) more resistant to diseases, and (4) easier to raise and keep in confinement; (5) hacking was well-developed long before its use for translocation; and (6) people who work with raptors have “green thumbs” and are fanatically devoted to their projects. Ethical and societal concerns about translocation as a method of conservation have been unduly emphasized in the literature of the past 20 years and have little relevance to raptor projects. Translocation has proved to be important for the preservation of rare and threatened raptors in nature.

INTRODUCTION

Since my earlier review (Cade 1986), reintroduction or, in a broader sense, translocation as a method for restoring raptor populations has increased greatly. In 1984 I was able to summarize only some 15 examples of diurnal raptor reintroduction (see, also, Garcelon and Roemer 1988), but by now there have been well over 50 translocation projects involving more than 25 species. The Richard R. Olendorff Memorial Library at the Raptor Research Center, Boise State University, Idaho contains more than 1,000 catalogued references on raptor translocation. In recognition of the general, worldwide interest in reintroduction and related techniques as methods of conservation, the IUCN Species Survival Commission established a “Re-introduction Specialist Group” in 1989, under the chairmanship of Dr. Mark R. Stanley Price, to coordinate and guide efforts in reintroductions among the many scientists and managers involved and to establish and administer a set of guidelines for acceptable protocols. In addition, many European projects have long operated under the criteria for

assessing the suitability of translocation set forth by the UK Committee for International Nature Conservation under the Nature Conservancy Council of Great Britain (Anon. 1976; see also Cade 1986).

Although the term has been used to refer to several kinds of activities involving the release of animals into natural habitats, "reintroduction" has a specific meaning, namely: "the re-establishment of a species by the release of individuals in an area where it naturally occurred but has been extirpated, usually through human influences" (Cade 1986). Since that is only one of several kinds of human intervention involving the release and establishment of animals in outdoor environments, a more inclusive term has been needed to cover all activities involving the manipulation of individuals to establish or bolster wild populations—including such actions as "restocking" or augmenting existing populations, island marooning (Williams 1977), and introduction into previously unoccupied range. The term "translocation" as used by the IUCN (1987; see also Griffith *et al.* 1989) serves that need: "Translocation is the intentional release of animals to the wild in an attempt to establish, reestablish, or augment a population." Thus, it covers such micro-manipulative techniques as "sibling rescue" (Meyburg 1978) and "fostering" (Cade 1978) to major introductions of species outside their natural ranges.

In the past 20 years there has been an increasing number of reviews and discussions about the technical effectiveness of translocation for species restoration and also about the ethical desirability of such methods, particularly as compared to more inclusive ("holistic") approaches such as habitat preservation and ecosystem management (see, for example, Myers 1979, Pitelka 1981, Csuti *et al.* 1987, Hutto *et al.* 1987, Newton 1988, Griffith *et al.* 1989, 1990, Povilis 1990, Varner & Monroe, 1990, Cade & Temple 1995, Snyder *et al.* 1996, and Wolf *et al.* 1996). My purpose here is to review the work on translocating raptors against this background of discussion about technological effectiveness and ethical validity and to show how translocation complements other methods of conservation. As a technique how successful have raptor translocations been in establishing or significantly augmenting viable, self-sustaining populations in nature? Do artificially established raptor populations fit naturally into their ecological communities and function as their wild counterparts do or did?

METHODS

To answer these questions I have reviewed much of the literature and unpublished reports on translocations. In addition, I sent out a questionnaire to 32 workers known to be involved in one or more raptor translocations and received usable replies from 31 of them. From these sources I obtained information on 52 translocation projects involving 25 species of diurnal birds of prey. There are a number of other translocations involving diurnal raptors, but these 52 projects are the ones for which I could obtain sufficient information for an evaluation, and they represent the spectrum of programmes and techniques being used.

I have not analyzed these data statistically, because each translocation represents a complicated synergism of details about species, location, time, methods, and people. When such complex variables and interactions are reduced to a few numbers for statistical analysis, I believe important and probably essential perspectives for evaluating success or failure get lost. Although it may be tedious and old-fashioned to analyze projects on a case by case basis, as Cade and Temple (1995) did, I believe that one often ends up with a better understanding of why some projects succeed while others fail than when projects are reduced to unidentifiable numbers that are then fed into high-powered computer programmes, the workings and significance of which many researchers have to accept on faith.

RESULTS

The data are summarized in Tables 1, 2, and 3. I judged 28 of the 52 projects to have been carried out with sufficient scope (based on number of birds released, area of releases, time, monitoring effort) to warrant evaluation of their success in establishing a self-sustaining population (Table 1). By "self-sustaining" I mean that given the current characteristics of the population (its size and

Table 1 Projects carried out with sufficient scope to warrant evaluation of success or failure.

Project	Area (Km ²)	No. Years	No. Rel. per Year	Age	Captive or Wild	Years Breeding	No. Pairs (Yr.)	% Suc.	Release Method	Habitat	No. Workers	Remarks
Peregrine—Eastern USA	>10 ⁶	20	>1300	J/n	C/w	19	132 ('97)	>60	H/f	A	10 ²	Young produced in 1997 = 185.
Peregrine—Midwest USA	>10 ⁶	15	>850	J	C	12	95 ('98)	c. 60	H	P/a	10 ²	Estimated 1/2 wild population derived from released birds.
Peregrine—Rocky Mts. USA	>10 ⁶	20	2112	J/N	C/w	17	438 ('97)	c. 80	H/f	P	10 ²	Some cross-fostering: estimated >1/2 wild pop. derived from released birds.
Peregrine—California	>10 ⁶	15	754	N/J	C/W	17	140 ('97)	c. 50	F/H	A/p	10 ²	Note: first captive-bred & fostered bird nested at Chipp River, n. Alberta, 1977.
Peregrine—So. Canada	>10 ⁶	20	1550	J/n	C	17	105 ('98)	> 50?	H/f	A/p	10 ²	First tree-nesting pairs in 1996-98.
Peregrine—Germany	>10 ⁶	20	803	J/N	C	15	>70 ('98)	>50?	H/f	P/A	Many?	Still increasing and considered self-sustaining.
Peregrine—Sweden (Norway)	>10 ⁶	16	>419	J/n	C/w	15	48 ('97)	74	H/f	P	10 ²	Breeding success = 72% in East, 58% in West; 3 pops.
Mauritius Kestrel	>10 ²	10	350	J/N	C/W	12	>100 ('97)	c. 65	H/f	P/A/M	1	Max. no. pairs in 1980s; currently estimated <5 pairs.
Seychelles Kestrel	40	1	13	A/J	W	19	>10 ('80s)	?	M	A	1	Breeding success ranged from 0 to 80%.
Lesser Kestrel—NE Catalonia	10 ⁴	9	378	J/n	C/w	8	12 ('98)	38	H/f	A	c. 10	Breeding success ranged from 27 to 100%.
Lesser Kestrel—W Catalonia	>50	8	329	J/n	C/w	5	16 ('97)	72	H/f	A	c. 10	First bred at 4 years in 1980.
Bald Eagle—New York State	>10 ⁴	13	213	J/n	W/c	19	40 ('98)	66	H/f	P	10 ²	Wild eggs hatched in lab, and chicks raised to 8 weeks.
Bald Eagle—So. Cent. USA	10 ⁶	8	275	J	W/c	6	>30 ('98)	71	H	A	c. 4-5	Reproduction above state average.
Bald Eagle—Calif. Coast Ra.	10 ⁴	9	66	J	W/c	8	7 ('98)	?	H	A	?	*Attempted breeding—none successful.
Bald Eagle—Catalina Island	10 ³	19	57	J/n	C/W	12*	3 ('97)	0	H/f	M	4	*Total production = 89 young fledged.
Bald Eagle—Massachusetts	>10 ³	7	41	J	?	10*	9 ('98)	?	H	A	?	18 of 30 occupied territories produced 34 fledglings in 1997.
Bald Eagle—Tennessee	10 ⁴	19	>269	J	C/W	16	37 ('98)	>50	H	P	?	11 of 19 occupied territories produced 20 fledglings in 1998.
Bald Eagle—Indiana	10 ⁴	5	73	J	W	7	19 ('98)	59	H	A	?	46 more released in 1993-1997.
White-tailed Eagle—Scotland	10 ⁴	11	82	J/I	W	14	c. 14 ('97)	c. 42	H	A	?	Some wild birds attracted to settle.
Osprey—Pennsylvania	10 ⁴	12	236	J	W	12	c. 35 ('97)	80	H	P	10 ²	In 1998, 91 pairs produced 171 young, but not all are released birds.
Osprey—Tennessee	10 ⁴	10	165	J	W	c. 15	>97 ('98)	>90	H	P	?	High survival of releases and conspecific attraction = success.
Osprey—Twin Cities, Minn.	>10 ²	12	144	J	W	10	>15 ('97)	?	H	A	40	Falconry escapees, 1965-1980s and 25 deliberate releases.
Griffon Vulture—France	>10 ²	6	61	A/I/J	C/W	16	71 ('97)	c. 60	A/H	P	5	High survival and productivity = quick success.
Goshawk—Great Britain	10 ⁵	15-20	c. 250	J/A	W	>30	c. 400	?	Hd/h	P/A	10 ²	High survival and productivity = quick success.
Red Kite—South England	>10 ³	6	93	J	W	7	>51 ('97)	90	H	P	12	High survival and productivity = quick success.
Red Kite—North Scotland	>10 ³	5	93	J	W	6	23 ('97)	>75	H	A	c. 10	High survival and productivity = quick success.
Harris' Hawk—So. California	>10 ²	12	220	J/I/A	C/w	16	3-5 ('90s)	80	H/h/f	M	c. 10	Limited and marginal habitat = dispersal.
Montagu's Harrier—Catalonia	>25	10	198	J/n	C/w	?	2-5 ('90s)	?	H/f	A-M	c. 10	Weak philopatry results in few returns to release area.

Table 2 On-going translocation projects.

Project	Area (Km ²)	No. Years	No. Birds Released	No. Rel. per Year	Age	Captive or Wild	Years Breeding	No. Pairs (Yr.)	% Suc.	Release Method	Range	Habitat	No. Workers	Remarks
Aplomado Falcon So. Texas	10 ⁴	5	214	10-101	J	C	4	4 (' 98)	75	H	P	A	c. 20	24 birds released experimentally 1985-89 prior to current project. Releases spread: 1990, 1992, 1995, 1997.
Lanner Falcon Israel	10 ³	6	46	av. >7	J	C	0	0	0	H	P	A	?	
White-tailed Eagle Israel	10 ³	5	13	c. 3	J	C	?	?	0	H	P	M	?	Poor habitat and small number of releases make project problematic.
Andean Condor Colombia	10 ⁴	9	42	c. 5	J	C	3	3	?	H	C	P	>10	Sexual maturity at 6-7 years.
Andean Condor Argentina	?	1	7	7	J	C	0	0	0	H	P	P	c. 5	Project just starting in 1997.
California Condor So. Calif.	10 ⁴	>6	39	2-8	J	C	0	0	0	H	C	P	19	11 removed; 19 remain in wild June 98; some pairing started.
California Condor Arizona	>10 ⁴	<2	19	6-13	J	C	0	0	0	H	P	P	5-6	15 birds survive to Aug. 1998; 2 dead, 1 lost, 1 returned to captivity.
California Condor Cent. Calif.	>10 ²	1	10	5+5	J	C	0	0	0	H	C	P	?	First 5 removed because of behavioural problems; 6-8 yrs to sex maturity.
Griffon Vulture Israel	10 ³	5	41	7-8	A/J	C/W	4?	>2? (' 97)	?	A?	P	A?	?	Some released birds have joined a wild colony 100 km away.
Bearded Vulture Euro. Alps	10 ⁴	12	72	av. 6	J	C	2?	2 (' 97)	50	H	C	P	10 ²	Good survival of released birds, 38-43 of 65 fledged up to 1996.
Euro. Black Vulture Mallorca	10 ³	9	33	9-10	A/J/Jn	C/w	6	10 (' 97)	50	H/A	P	A	>10	Last release in ' 92, bred 9 years after 1st release; total pop.= 60 birds.
Euro. Black Vulture France	>10 ²	7	26	av.>3	A/J/Jn	C/w	2	4 (' 98)	25	H/A	P	A	>10	Breeding after 4th yr; young adults will promote pop. growth.
Osprey England	c. 25	2	14	6-8	J	W	0	0	0	H	C	P	.10	First 2 yrs of 5 year project; 1 dead in Senegal., 12 migrated
Red Kite Midlands, England	>10 ²	3	49	16	J	W	2	4 (' 97)	c. 90	H	C	P	>10	On the way to successful establishment.
Red Kite Central Scotland	>10 ²	2	37	18-19	J	W	1?	2?	0	H	P	A	?	High survival of releases; attempted breeding in 1997 after first year.

Table 3 Experimental or casual translocation projects.

Project	Area (km ²)	No. Years	No. Birds	No. Rel. per Year	Age	Captive or Wild	Years Breeding	No. Pairs (Yr.)	% Suc.	Release	Range Method	Habitat	No. Workers	Remarks
Bat Falcon Guatemala	<10 ²	2	16	av. 8	J	C	0	0	0	H	P	M	1-2	11 of 16 young fledged and dispersed; none settled to breed.
European Hobby England	<10 ²	8	10	>1	J	C	0	0	0	F/H/t	C	A	1-2	10 young from one captive pair successfully released.
Madagascar Fish Eagle	10 ²	3	9	av. 3	N	W	0	0	0	SR	C	P	2-3	7 survived and increased productivity from 0.15 to 0.28 per pair.
Osprey No. Scotland	<10 ²	1	4	4	J	W	0	0	0	H	P	A	1?	Late hatch of fledged juveniles in September, 1979.
Spanish Imperial Eagle	>10 ²	7	12	1-2	N	W	0	0	0	SR	C	A?	?	8 of 12 fledged successfully (see text for significance).
Golden Eagle Sweden	>10 ²	13	25	c. 2	J	C	?	1?	?	H/t	P	M?	1	All offspring of one captive pair.
Andean Condor Peru	>10 ³	1	12	12	a/J/N	C	0	0	0	H/hd	C	A	c. 5-6	7 of 11 captive bred survived >2 yrs at termination; good integration with wild birds.
Common Buzzard England	c. 570	5	44	5-17	J	W/C	2	3 ('97)	100	H	C	P/A	11	Exptl. study: high dispersal reduces settlement in release area, but dispersal and survival normal for the species.

Footnotes to Tables 1-3.

Key to symbols used. Upper case letters mean primary use; lower case mean secondary use. Age: A = adult, I = immature (more than 1 year old, J = juvenile (including fledglings), N = nestlings. Release method: A = use of acclimation cage, F = fostering, H = hacking, Hd = hacking, M = island marooning. Range: C = central, P = peripheral. Habitat: P = prime, A = average, M = marginal.

numerical trend, survivorship, breeding success, contact with other populations) and the prevailing environmental conditions, the population should persist indefinitely, barring some unpredictable catastrophic event (see Mace & Lande 1991 for non-parametrical but objective ways to assess population viability). I have not used population viability analysis or any other probability model (see Beissinger & Westphal 1998 and Reed *et al.* 1998 for the limitations and pitfalls in using such models), although such models have been applied to specific raptor translocations, as indicated beyond. For 15 on-going projects only an interim evaluation is appropriate (Table 2), while nine others were either experimental in nature or lacked the scope required for success (Table 3). The projects are further described in the following sections by grouping them according to the main functional (or phylogenetic) categories of diurnal raptors—falcons, eagles, fish-hawk, Old World vultures, condors, and hawks *sensu lato*.

Falcons

1. Peregrine Falcon *Falco peregrinus*, Table 1—Although there were a few half-hearted attempts to translocate raptors in the early decades of this century and even before, the first coordinated and systematic efforts involved the Eagle Owl *Bubo bubo* in Germany and Sweden beginning in the 1960s and 1970s (Cade 1986). Also, about the same time, a partly organized and partly clandestine effort to release Fennoscandian Goshawks *Accipiter gentilis* resulted in the reestablishment of that species in Great Britain (Marquiss & Newton 1982, Cade 1986). It was, however, the plight of the Peregrine on two continents that called forth the largest, most sustained, and coordinated programmes to restore an endangered species by captive breeding and translocation (Hickey 1969, Cade 1988a, Cade *et al.* 1988, Cade 1990, Enderson *et al.* 1995, Cade *et al.* 1997, Cade 1998).

The idea to breed Peregrines in captivity and to release the progeny back into the wild developed at the International Peregrine Conference held at the University of Wisconsin in 1965 (Hickey 1969, Cade 1988a), consequent to the realization that the species was severely threatened on two continents. There have been seven major, regional programmes, five in North America and two in Europe (see Table 1), as well as several other smaller or recently started projects, particularly in Europe.

In 1970 a programme of research and management to reintroduce captive-bred Peregrines into the vacant range of the eastern United States began at the Laboratory of Ornithology, Cornell University—soon to become known as The Peregrine Fund (Cade & Temple 1977). In 1974 The Peregrine Fund established a second breeding facility and release programme for the Rocky Mountain region at Fort Collins, Colorado, in association with the Colorado Division of Wildlife (Burnham *et al.* 1988). Later, in 1984, a permanent headquarters for the entire organization became established in Boise, Idaho at a facility known as The World Center for Birds of Prey, under the leadership of William A. Burnham.

In 1972 the Canadian Wildlife Service established a falcon breeding centre at Fort Wainwright, Alberta under the supervision of Richard Fyfe, to produce the endangered *anatum* Peregrine for reintroduction into southern Canada where, as in the eastern United States, the species had been totally extirpated (Fyfe 1976, 1988). In California, a Pacific Coast programme—the Santa Cruz Predatory Bird Research Group—got underway at the University of California, Santa Cruz around 1975. Originally founded by James C. Roush III, it has been directed by Brian J. Walton since 1977 (Thelander & Walton 1988). The fifth programme, in the upper Mississippi River valley and western Great Lakes region (Midwest USA), began in 1982 under the leadership of Patrick Redig at the Raptor Center and Harrison B. Tordoff at the Bell Museum of Natural History, University of Minnesota (Redig & Tordoff 1988).

Guided by endangered species recovery plans approved by the U.S. Department of the Interior, Fish and Wildlife Service and Canadian authorities, and involving the collaboration of dozens of federal, state, and provincial agencies, numerous non-governmental organizations, and more than 1,000 individual workers, these programmes have resulted in the release of more than 6,000 Peregrines across the North American continent since 1974 (Enderson *et al.* 1995, 1998). They unquestionably constitute the largest, coordinated effort ever mounted to recover an endangered species. Most of the principals involved were or are falconers, and the influence of falconry on the success of these programmes needs to be acknowledged (Cade 1988a).

The release of Peregrines by hacking (Sherrod *et al.* 1981) began in the eastern USA in 1974. By 1980, when some 204 falcons had been released in nine states, the first three established pairs nested successfully, and that year fostering of captive-bred chicks into wild nests was added as a second method of release (Barclay & Cade 1983). Large scale releases by hacking and fostering continued through the 1980s up to 1992; at the same time the number of established breeders grew at rates of 25 per cent per annum or more (Barclay 1988, Enderson *et al.* 1995). By 1991 when the major release effort ended, more than 1200 captive-bred Peregrines had been released in 13 states and the District of Columbia, at least 91 territorial pairs had been established, and annual production of fledged young exceeded 100 birds. Since then the breeding population has continued to grow on its own (annual releases have averaged only a dozen or so birds) at an average rate of >7 per cent per year, yielding 132 territorial pairs counted in 1997 (Mesta 1998), slightly more than one third of the number estimated to have existed before the DDT-induced extirpation (Hickey 1942). More than 900 young fledged from 1992 through 1997 (data compiled by L. Kiff, The Peregrine Fund archives). About one third of the breeding pairs nest on natural cliff eyries, mainly in the mountains of northern New York State and New England; but most of the others are found in urban habitats (Frank 1994, Cade *et al.* 1996a) and on nesting towers in coastal salt marshes (Steidl *et al.* 1991, Therres *et al.* 1993).

The continuing rate of increase in population size, the current number of breeding pairs, and their rate of reproduction all indicate that this eastern population is now self-sustaining and will continue to grow to whatever the current carrying capacity of the eastern environment may be (Cade *et al.* 1997). In addition, this population is known to have exchanged dispersing individuals with the expanding Canadian population to the north and with the midwestern population in the USA, so that it is no longer a demographically or genetically isolated population (Barclay 1995, M. Martell *et al.* *in litt.* 1998).

Following two earlier attempts to hack Peregrines in habitats infested with Great Horned Owls *Bubo virginianus* along the upper Mississippi River, the midwestern programme got under way in 1982, using young, captive-bred falcons produced by private breeders, mostly falconers (Redig & Tordoff 1988). The first successful breeding of released falcons occurred in 1987, following the hacking of 102 young birds. The population grew rapidly at first with the continued annual release of 18 to >100 birds, most of the pairs settling in metropolitan and urban areas, where they nest on buildings, bridges, and power plant smokestacks (Cade *et al.* 1996a). The released birds failed to establish themselves as breeders at historical eyries along the Mississippi owing to depredation of the Great Horned Owl (Cade *et al.* 1989), but they have done so along the northern shore of Lake Superior and elsewhere where owls are not a problem. By 1998, after 857 Peregrines had been released in 12 states and two Canadian provinces, the known adult population consisted of 99 territorial pairs, 84 of which nested and produced 205 young (H. B. Tordoff *et al.*, unpubl. rept. 1998). A demographic analysis of this population, considering data on dispersal, survival of territorial adults, age of first breeding, longevity, lifetime reproduction, and fidelity to mate and territory from 1982-1995 indicates that this population is self-sustaining and capable of growing to the current carrying capacity based on the number of serviceable breeding locations in the region (Tordoff & Redig 1997). Even an earlier population viability analysis, carried out when this population consisted of only 52 territorial pairs in 1991, showed that under a variety of supposed conditions calculated extinction rates over 100 years varied from 0 to 5.2%, mostly <1.0% (Moen & Tordoff 1993). Also, because of its high reproductive capacity and long range dispersal of individuals, this population likely serves as a source for repopulating adjacent regions, particularly to the west where few Peregrines as yet have resettled in the northern Great Plains.

Restocking of Peregrines in the Rocky Mountain states and Pacific Northwest began in 1977 and ended in 1997. The total number of birds released by hacking and fostering (a few also cross-fostered to pairs of Prairie Falcons, *Falco mexicanus*) was 2,122 falcons. Released falcons were known to begin breeding in Colorado by 1980, and by 1997 the total number of known pairs in the eight states of Washington, Oregon, Idaho, Montana, Wyoming, Colorado, Utah, and New Mexico was 438, whereas the known number before restocking started was only 21 pairs (Burnham 1998).

The number of reported pairs must be taken as a minimum figure, because it is now difficult for field workers to survey all potential nesting locations in these states in a given year.

It is also difficult to know how much of the increase in breeding numbers can be attributed to restocking and how much to natural increase, as both processes occurred simultaneously; but with so few wild pairs present at the start, restocking must have accounted for most of the population growth, particularly in the early years. For example, in the three states of Idaho, Montana, and Wyoming there was but one known wild pair in 1980 when restocking started. In 1997 these states held a minimum of 77 pairs; in the early years of reestablishment all closely observed adults at eyries wore bands identifying them as captive-bred and released birds. Similarly, in Colorado, perhaps the most intensively monitored state in the West, there were only six known pairs remaining when restocking started in 1977; after the release of 405 young falcons through 1990, the number of breeding pairs had grown to more than 80 in 1996 and 1997, three times the number known in Colorado before the DDT-induced decline (Enderson *et al.* 1988, Enderson *et al.* 1995, Mesta 1998). Again, during the early years most of the breeders wore bands identifying them as released birds (Cade *et al.* 1997).

Fewer than 10 surviving pairs of Peregrines were known in California before restocking began in 1977 (systematically in 1980), whereas more than 200 historical eyries had been catalogued for the state (Walton & Thelander 1988, Walton *et al.* 1988b). Using hacking, fostering, and cross-fostering with Prairie Falcons, and young hatched from salvaged, thin-shelled wild eggs as well as captive-produced young, Walton and his associates released 754 falcons in California through 1997 (Walton 1997). By 1992 there were 116 occupied breeding territories in the state. Although no statewide count has been made since then, continued increases in several regions and an extrapolation of the known population growth from 1975 to 1992 indicate that there were around 140 pairs in the state in 1997, and the prediction is that the California Peregrines will exceed 200 pairs before reaching a limit imposed by environmental carrying capacity (Walton 1997), contrary to the population model developed by Wootton and Bell (1992).

From 1976 through 1996, Canadian workers released 1,550 Peregrines, mostly by hacking, in southern Alberta, s. Saskatchewan, s. Manitoba, s. Ontario and Quebec, and in the Maritime provinces, a region in which only one wild pair is known to have persisted until 1980. By 1988, after 563 young falcons had been released in 24 areas of southern Canada, only 12 nesting pairs could be located in the entire region, and the programme underwent re-evaluation (Holroyd & Banasch 1990). One reason for the apparent low success rate might have been high mortality related to long distance migration, but a more likely reason has to do with the great expanses of wild country in Canada and the limited number of observers available to find Peregrines. Following recommendations that more young (20-30) be released per area per year and that a more intensive monitoring scheme to locate nesting pairs be instituted, by 1995 there were 41 known nesting pairs in southern Canada out of 100 sites surveyed; 32 unsurveyed sites could have been occupied by an additional 12-13 pairs (U. Banasch *in litt.* 1997). In 1998 the number of known occupied sites had climbed to 105 for southern Canada excluding British Columbia (G. Holroyd *in litt.* 1998, Mesta 1998). Although this population is thinly spread over a vast landscape, it is now growing rapidly, and it probably receives some wild recruits from Labrador, the northern tier of states, and from northern Alberta.

Owing in part to the success of these regional translocations and in part to natural recovery, the U. S. Fish and Wildlife Service published a formal notice to remove the American Peregrine Falcon *F. p. anatum* from its list of threatened and endangered wildlife (Mesta 1998). The final notice delisting this subspecies is expected in 1999.

The German programme has been largely a private initiative of the Deutscher Falkenorden carried out under the general supervision of Prof. Dr. Christian Saar, with help from other groups and government agencies (Saar 1988, *in litt.* 1998). The effort has been concentrated in regions north of the River Main in both former West Germany and East Germany where the Peregrine had been totally extirpated by pesticides in the 1960s, with only a small population of some 70 pairs remaining to the south in Bavaria and Baden-Württemberg (Mebs 1988). Young falcons produced in captivity by falconers, mainly at Saar's Wanderfalken-Zucht-und-Forschungsstation in Berlin (and later,

Hamburg) were released by hacking, fostering, and cross-fostering. Releases began in 1977 and have continued to the present. By 1997, 803 Peregrines had been released and a minimum of 70 territorial pairs established in Northern Hessen, Lower Saxony, Schleswig-Holstein, Hanover, Berlin, Brandenburg, Mecklenburg-Vorpommern, Saxony, and Thüringen. In addition, an unknown number of released Peregrines dispersed southward and joined the expanding wild populations in Bavaria and Baden-Württemberg, but so far no southern, cliff-nesting falcons have dispersed northward. The reintroduced falcons are nesting mostly on cliff sites or on man-made structures in urban habitats and on offshore lighthouses in the North Sea (Saar 1988). Work to establish cliff-nesting Peregrines ended in 1996 with a project in Sächsische Schweiz National Park, where 12 pairs are now nesting (C. Saar *in litt.* 1998).

The remaining goal in northern Germany is to reestablish the tree-nesting population of Peregrines that formerly lived in the low country adjacent to the Baltic Sea, a population that extended through northern Poland and into the Baltic states. This task is being accomplished by hacking from artificial nests in trees and by cross-fostering young Peregrines with Goshawk parents. The first tree-nesting pair bred in a natural tree nest in 1996 and has raised three young each year through 1998; two other unsuccessful pairs have been reported (Langgemach *et al.* 1997, C. Saar *in litt.* 1998). A similar project started in northwestern Poland in 1990 in the expectation that birds from the two sources will link up to establish a viable population across the border of the two countries (Wisniewski 1995, Trommer 1997).

The Swedish project has been supervised by Peter Lindberg at the University of Göteborg and involves the Swedish Society of Nature Conservation, Swedish Environment Protection Board, and a number of other organizations, and many individuals, including some in Norway. The original goal was to restock a region of some 15,000 km² in SW Sweden where only a few pairs remained, until a breeding population of 30 pairs (half the pre-1955 number) had been achieved (Lindberg 1988). Release of captive-bred falcons began in 1982 and continues to the present, with a total of 419 young hacked or fostered into nests of wild Peregrines. By 1997 there were 23 pairs in SW Sweden and another 25 pairs in SE Norway just across the border (Lindberg 1997, *in litt.* 1998). The project has taken longer than originally anticipated in part because of depredations on the young falcons by reintroduced Eagle Owls.

Other small, lesser known or just starting projects to restore Peregrines are underway in various European countries. The idea, apparently promoted by the Wisconsin Peregrine Society, to establish nesting Peregrines in cities is especially popular (e.g., Warsaw, Luniak 1995).

2. Mauritius Kestrel *Falco punctatus*, Table 1—Several other species of falcon have been the subjects of translocation projects. Certainly the most outstanding has been the programme to restore the Mauritius Kestrel (Jones 1980, Cade & Jones 1993, Jones *et al.* 1995), perhaps the most significant rescue of any endangered species so far effected by captive breeding and translocation. Carried out under the direction of Carl Jones since 1979 and supported by the Government of Mauritius, the Mauritius Wildlife Fund, Jersey and International Wildlife Trusts, and The Peregrine Fund, the programme released 350 birds by hacking and fostering in the years from 1984 to 1994, using both captive-bred kestrels and birds hatched from wild eggs. The population on the island has grown from only two known pairs in 1974 (Temple 1977) to more than 100 pairs in 1997 distributed in three geographic locations covering much of the 30,000 hectares now estimated to be suitable habitat, and it seems likely that the population will eventually grow to 200–250 territorial pairs and several hundred total individuals (Jones *et al.* 1995, C. Jones, *in litt.* 1998).

3. Seychelles Kestrel *Falco araea*, Table 1—Jeff Watson's (1989) project with this species on Praslin Island is unique among the raptor translocations considered here in that it consisted of a single, hard release by island-marooning of three immature and 10 adult kestrels trapped on Mahé and released within two days on the smaller island (4,000 hectares), where the species had been extirpated following extensive deforestation and fires in the first half of the 20th Century. This project probably belongs in the experimental category, but I have included it in Table 1 because it was a deliberate attempt to reintroduce a population by the hard release of a minimum number of individuals.

It was thought that by the 1970s sufficient regrowth of forest had occurred to provide suitable

habitat once again. Kestrels began nesting soon after release, and by 1980 at least 10 pairs were present. Unfortunately there has been no systematic, island-wide survey since 1980. Casual observations by staff of the Conservation Division on Praslin indicate that the early momentum of population growth has not been sustained, as there appear to be no more than two or three pairs remaining, perhaps owing to the use of predator-accessible nest sites in coconut palms and on buildings (J. Watson *in litt.* 1998). A thorough re-survey needs to be done.

4. Aplomado Falcon *Falco femoralis*, Table 2—Following a few experimental releases in 1985–1989, The Peregrine Fund began a large scale, continuing project to release captive-bred Aplomado Falcons by hacking in south Texas (Laguna Atascosa National Wildlife Refuge and adjacent private properties), in cooperation with the U. S. Fish and Wildlife Service and the Texas State Parks and Wildlife Department (Cade *et al.* 1991). The northern subspecies *F. f. septentrionalis* is listed as endangered in Mexico, and the southwestern USA where it had been extirpated as a breeding species by the 1940s and 1950s, probably owing to habitat changes. Suitable savanna-like habitat appears to have returned to southern Texas following brush-removal programmes and lowered chemical contaminant levels in the environment (Mora *et al.* 1997). Since 1993, more than 300 young Aplomado Falcons have been hacked in south Texas, more than 200 of them in the last two years. The first established pair bred in 1995, and there have been four or five pairs located on and around the Laguna Atascosa refuge in 1996–98 (P. Jenny *in litt.* 1998), but much suitable habitat remains to be surveyed. Many single birds are also present. Predation by Great Horned Owls and other predators has been a problem (Perez *et al.* 1997), and there could be a scarcity of suitable nest sites (old stick nests of other hawks and ravens); but it appears that successful establishment of a viable breeding population is just a matter of continuing to release birds for a few more years. An intensive search in 1999 located >15 territorial pairs, (W. Heindrich *in litt.*).

5. Lesser Kestrel *Falco naumanni*, Table 1—In Catalonia, Spain, where the species last bred naturally in 1986, there are two projects to reestablish nesting colonies of the Lesser Kestrel, mostly by hacking but also by fostering and cross-fostering to Common Kestrel *Falco tinnunculus* parents; about 95 per cent of the kestrels were produced in captivity (Pomarol 1993). These projects are carried out under the direction of the wildlife service of Catalonia (Sevei de Protecció i Gestió de la Fauna, Direcció General del Medi Natural) and the supervision of Manel Pomarol.

At the site in Alt Emporda county, 378 kestrels have been released since 1989, and between 70 and 76 per cent have survived their first year. Two pairs first bred in 1990, and by 1998 there were 12 pairs in the colony; but annual breeding success has varied from 0 to 80 per cent, averaging only 38 per cent. Problems that retarded population growth but that have now been corrected include (1) a rat poison used in the building where the kestrels are nesting, killing half the chicks and breeders in 1993, (2) high predation of adults and chicks by the marten (*Martes foina*) in 1994–95, and (3) a vineyard pesticide that caused reproductive failure in 1997.

At the site in Noguera county, 329 birds have been released since 1993, and there were 16 established pairs in 1997. Their annual breeding success rate has varied between 27 and 100 per cent, averaging 72 percent over five years. Only one serious problem of mammalian predation occurred in 1994 and 1995.

Despite the high mortality, these colonies have grown slowly over six to seven year periods. Continued release of birds for a few more years, combined with providing the colonies with superior, manufactured nest-sites (Pomarol 1996), should result in self-sustaining, even expanding populations (M. Pomarol, *in litt.* 1998).

6. Lanner Falcon *Falco biarmicus*, Table 2—There is an on-going project at Mount Carmel, carried out by the Society for the Protection of Nature in Israel, to establish breeding Lanners in the area by hacking captive-produced fledglings (Getreide & Hatzofe 1991). A total of 45 falcons has been released in 1990, 1992–95, and 1997. Returning birds have been seen around Mount Carmel, but so far no breeding has been observed. Releases are continuing (O. Hatzofe *in litt.* 1998).

7. Other *Falco* spp.—There have no doubt been several other exploratory attempts to translocate falcons. After all, falconers have been hacking birds for a thousand years or more, and enough falcons have escaped during hack to have suggested to more than a few minds that the process might be used to establish wild breeding pairs in new or unusual locations.

J. P. Vannini (1989, 1990), Fundacion Interamericana de Investigacion Tropical, in cooperation with the Comision Nacional de Areas Ambiente, hacked 16 captive-produced Bat Falcons *Falco albicularis* in mixed degraded forest and agricultural lands on the Pacific Versant of Guatemala, where the species is now virtually extirpated. Post-hacking survival was good (11 of 16 birds), but no pairs became established, possibly because the environment is no longer suitable (Table 3). In Devon, England, Heather Woodland (1993) bred a pair of Hobbies *Falco subbuteo* in captivity and released 10 of their young by hacking, fostering, and one trained by a falconer before release (Table 3). She concluded, correctly I think, that hacking was the preferred method, the one least likely to have unintended, adverse side effects.

Eagles

1. The Bald Eagle *Haliaeetus leucocephalus*, Table 1—This species has been the subject of numerous translocation projects in the United States since the first efforts by The Peregrine Fund in New York State in 1976 (Cade 1983, Nye 1988). By 1985, Nye (1988) was able to summarize information on the hacking and occasional fostering of 390 Bald Eagles in 13 states and the province of Ontario, Canada. Unfortunately, there has been no continent-wide assessment of all these projects since then, but well over 1,000 eagles have been translocated in various parts of the country. The seven projects summarized here show the range of accomplishments.

The Peregrine Fund hacked the first Bald Eagles (wild eaglets from Wisconsin) in New York State in 1976 and 1977 under contractual arrangements with the State Department of Environmental Conservation and the U. S. Fish and Wildlife Service (Milburn 1979). At that time there was only one poorly reproducing pair in the state. Afterward, the state's Endangered Species Programme took over the project under the leadership of Peter Nye, and through 1988 the state hacked a total of 198 eaglets (mostly from Alaska) at four different locations (Nye 1990). The first two eagles hacked at the Montezuma National Wildlife Refuge in 1976 were found paired and breeding at a nest in Jefferson County about 135 km north of their release site, where they produced young nearly every year well into the 1990s. Although the original birds are gone, the territory remains occupied by a breeding pair (P Nye *in litt.* 1998).

The nesting population in New York grew steadily, and by 1998 there were 40 territorial pairs, and the total production of young for all years through 1997 had exceeded 230 fledglings (P. Nye *in litt.* 1998). In addition, a number of the released eagles dispersed into other states and Canada (Nye 1988). This has been the best documented and most successful of the various Bald Eagle translocation projects, and it demonstrates the important point that it is easier to translocate and establish a viable population of long-lived birds with a high survival rate, even with delayed sexual maturity and a low reproductive rate, than it is to work with so-called *r*-selected species. The latter require a larger number of releases over a shorter period of time to be successful.

The George M. Sutton Avian Research Center in Oklahoma adopted a variation of the hacking procedure by obtaining eaglets for release from wild eggs collected in Florida from nests containing 3 eggs (Sherrod *et al.* 1987, 1989). Eaglets hatched from artificially incubated eggs were reared in captivity to 8 weeks of age before placement at hack. Between 1985 and 1992, 275 eaglets were successfully hacked at 15 locations in Oklahoma, Mississippi, Alabama, Georgia, and North Carolina. By 1991 some released eagles were nesting (Jenkins & Sherrod 1993), and by 1997 there were 26 pairs known in Oklahoma; in Georgia the population increased from 17 pairs in 1993 to 31 pairs in 1997; in Alabama 9 or 10 pairs consisted of one or more released birds; and in Mississippi the population had grown from 8 pairs in 1993 to 13 in 1996 (Jenkins 1996, *in litt.* 1998).

Unfortunately, thorough monitoring of post-release results has not been possible throughout the region of releases, and the wild eagles have been increasing at the same time; but it seems likely that released birds have accounted for a good part of the population growth. Meyers and Miller (1992) found no difference in initial hacking success between eaglets reared at the Sutton Center and wild-taken eaglets, but the captive-reared eaglets showed a tendency to remain near the hack-site for a longer time after fledging than the wild birds and to feed more on artificially provided food. These authors concluded that the captive-reared birds might have developed a stronger site tenacity for the

release location by remaining longer and also may have had a higher first year survival by feeding longer on provided food before dispersal.

The Ventana Wilderness Sanctuary Research and Education Center in Monterey, under the direction of Glenn Stewart, initiated a project in 1986 to establish Bald Eagles in the Central California Coast Recovery Zone by hacking wild-taken eaglets at a site on the coast near Big Sur (Lucido & Sorenson 1997). In all 66 eaglets have been released through 1997, and seven territorial pairs are known to be established in 1998, ranging from Alameda County in the north to Santa Barbara County in the south (K. Sorenson *in litt.* 1998). It is likely that more intensive searching in this extensive range would yield additional pairs. Although the project was intended to establish nesting pairs along the coast, so far all breeding pairs have gone inland to rivers and reservoirs in the mountains of the Coast Range, probably an adaptive choice considering the results of the next project.

On Santa Catalina Island, off the coast of California from Los Angeles, David Garcelon and associates of the Institute for Wildlife Studies, Arcata have been trying to reestablish Bald Eagles for 19 years, in association with the Santa Catalina Island Conservancy (Garcelon 1988, Garcelon *et al.* 1989). Using both captive-bred and wild-taken eaglets, they have hacked and fostered 57 birds on the island. Survival of translocated eagles has been good, and one or more pairs (three from 1991 to 1997) have been attempting to nest on the island for 12 years (Garcelon *et al.* 1995); but their eggs fail to hatch owing to high residue levels of DDE in the eagles' food, mainly gulls that forage in waters where organochlorine residues remain high (Garcelon *et al.* 1989, Jenkins *et al.* 1994). On the other hand, the established pairs are perfectly able to rear fostered young (D. K. Garcelon *in litt.* 1998). In this case, a technique for translocation that worked well has failed to reestablish a viable nesting population because the environment is still not suitable for breeding eagles.

In Massachusetts, a state where only two or three pairs of Bald Eagles are known to have bred prior to c. 1905 (Forbush 1929), a translocation project began in 1982 at the Quabbin Reservoir in the central part of the state (Nelson 1986), a joint effort of the State Wildlife Division and the U. S. Fish and Wildlife Service. Through 1988, 41 eaglets were hacked in Massachusetts: the first pair nested in 1989, and in 1998 there were nine territorial pairs, five of them on the Quabbin Reservoir, three along the Connecticut River, and one in the southern part of the state; a tenth pair settled in New Hampshire (T. French, pers. comm. 1998).

In 1980, a major translocation project began in Tennessee in the Land Between the Lakes and spread elsewhere, a joint effort of the State Wildlife Resources Agency, U. S. Fish and Wildlife Service, Tennessee Valley Authority, and the National Foundation to Protect America's Eagles (Hammer *et al.* 1983a and b, Hatcher 1983, 1991). No eagles were known to be nesting in the state between 1961 and 1983. Through 1997, 269 eaglets had been released at seven hack-sites. Only captive-bred birds have been used recently. The first pair formed in 1983 and produced one young; two other pairs are known to have bred at three years of age (R. M. Hatcher pers. comm. 1998). In 1997 there were 30 occupied territories, 18 of which produced 34 fledged young; in 1998 there were 37 occupied territories, not all of them by eagles hacked in Tennessee (R. M. Hatcher pers. comm. 1998).

From 1985 through 1989 the Indiana Department of Natural Resources conducted a Bald Eagle translocation project at Lake Monroe in the central part of the state, supported entirely by the Nongame Wildlife Fund through a citizens' income tax checkoff (Castrale 1991, Iverson 1991, Castrale & Parker 1998). Eagles were last reported to nest in Indiana in 1897. Seventy-three eaglets from Alaska and Wisconsin were hacked at the lake. The first territory became occupied in 1988, the first nesting attempt occurred in 1989, and the first two successful nestings in 1991. By 1998 there were 19 occupied territories, with 15 breeding attempts and 11 successful pairs raising 20 young. Not all of these breeders were eagles released in Indiana, as some wild birds and birds hacked in other states have dispersed into Indiana, just as Indiana's released eagles have dispersed as far west as the Mississippi River, north into Michigan and New York, and south into Texas. The Alaskan birds have ranged significantly farther from Monroe Lake than those from Wisconsin, but there was a decided tendency for all the birds to range closer to home, under 160 km, as they reached an age of more than three years (Castrale 1991).

It should be noted that many of these Bald Eagle translocation projects have taken advantage of new habitat created by dams and artificial reservoirs, so that eagles are now nesting in some localities where they had not occurred historically. Also, as with the Peregrine Falcon, the contribution of translocation to the recovery of regional eagle populations has sometimes been obscured by a simultaneous increase in the number of wild birds; but an interesting phenomenon, noted particularly in Indiana and Tennessee, has been the clustering of new nests and territories in localities where hack-towers were situated. In part this clumping no doubt reflects the well known philopatry of this species, but since not all of the eagles nesting in these areas are released birds, it must also reflect conspecific attraction (Stamps 1988) of wild eagles into locations frequented by released birds (Iverson 1991, B. Hatcher pers. comm. 1998).

2. White-tailed Sea Eagle *Haliaeetus albicilla*, Tables 1 and 2.—In Europe there has been interest in using translocation to reestablish breeding sea eagles in parts of the range where the species has been extirpated or greatly diminished in number (Love & Ball 1977, Fentzloff 1984, Cade 1986). The best documented and most successful has been the project in Scotland carried out initially by the Nature Conservancy Council and subsequently by the Scottish Natural Heritage and the Royal Society for the Protection of Birds (Evans *et al.* 1994), with the field work in the early years under the supervision of John A. Love (1983). From 1975 through 1985, 85 eaglets from Norway were hacked from cages (a few were released in other ways) on the Isle of Rum (also Rhum) in the Hebrides. Survival was good, and the eagles dispersed widely over western Scotland, northern Ireland, and Shetland; however, all reestablished nesting locations have been within 100 km of Rum (Green *et al.* 1996). The first territorial pair was found in 1981, the first recorded young produced in 1985 (6 pairs on territories); 10 pairs were located in 1991, and by 1997 there were 13 to 15 territorial pairs and a total of 64 young fledged to the wild (Evans *et al.* 1994, Roy Dennis *in litt.* 1998).

An evaluation of the results of the programme through 1992 led to a long-term viability analysis of this reintroduced population (Green *et al.* 1996). Because the estimated population growth rate based on available demographic data had a 95% confidence interval that encompassed both a decline of 11% per year as well as an increase of 11% and because simulation models indicated that there was a probability of 0.6 that the present population would become extinct in 100 years even if all demographic parameters operated at their maximum-likelihood values, the managers concluded that a further release of 60 juveniles would substantially reduce the risk of extinction. An additional 46 young from Norway were released in the northwest Highlands in 1993-1997, and more are scheduled for 1998 (Roy Dennis *in litt.* 1998). This study shows the usefulness of demographic analysis in estimating the success of a translocation project.

At the Hula Nature Reserve and at Kfar Rupin in the northern part of the Rift Valley in Israel there is a small project to reestablish sea eagles (Ohad Hatzofe *in litt.* 1998). In 1992-95 and 1997, 13 eaglets were hacked. One or more pairs had formed by 1997, but no successful breeding has occurred as yet (Table 2).

3. The Madagascar Fish Eagle *Haliaeetus vociferoides*, Table 3—The young of this globally endangered species apparently engage in obligate siblicide or "Cainism", as two eggs are frequently laid but never more than one chick fledges (Watson *et al.* 1996, *in press*). Some time ago Meyburg (1978, 1983) carried out experiments and described a procedure now referred to as "sibling rescue" by which the reproductive output of such species can be increased by various human interventions to save the weaker chick, "Abel," and allow both birds to fledge. Watson *et al.* (1996, *in press*) used sibling rescue by constructing a "double nest" with a barrier between the two parts and returning Abel to one side of the nest at 4 weeks of age. They were able to increase the productivity of a local population of Madagascar Fish Eagles ($n = 10$ pairs) to 0.28 young per territory over a three year period, compared to a productivity of 0.15 young in an unmanipulated population. While this procedure had a significant effect on productivity, there are important unanswered questions about (a) the quality of fledglings when two chicks must be fed and reared simultaneously and (b) the influence of this increased effort on survivability of the parents. A better tactic might be to take the extra offspring from nests of two, hand-rear them, and then hack them out in unoccupied habitat in an attempt to increase the number of nesting pairs by increasing the distribution or by creating additional populations (Watson *et al.* 1996).

4. The Spanish Imperial Eagle *Aquila adalberti*, Table 3—Ferrer and Hiraldo (1991) also used sibling rescue to increase the productivity of this species in Doñana National Park, Spain. Broods of up to three chicks occur, and the young engage in facultative siblicide. The technique was to transfer chicks in nests with three young to nests with one (Meyburg & Garzón 1973), but no significant difference in productivity was detected between manipulated and unmanipulated nests, and in any case the main limitation on productivity was low hatching success, not survival of nestlings. Again, a better use of chicks from large broods would be to hack them out in unoccupied habitat.

5. The Golden Eagle *Aquila chrysaetos*, Table 3—In southern Sweden, Dag Peterson has released 25 captive-produced eaglets (22 by hacking after fledging in a cage and 3 by fostering), all from a single breeding pair. Survival to independence has been high, and possibly one breeding pair in southern Sweden originated from these releases (Peter Lindberg *in litt.* 1998).

6. The Harpy Eagle *Harpia harpyja*, Table 3—Considered a flagship and umbrella species for conservation of the neotropical forests, the Harpy Eagle has virtually disappeared from the northern portion of its original distribution from Panama into Mexico. Despite the effort to establish tropical forest reserves, many are too small to maintain more than a few, isolated pairs of Harpy Eagles. Captive propagation and translocation will be necessary to maintain the genetic diversity of these isolates and to mitigate stochastic and catastrophic losses of individuals in these small and increasingly isolated populations. For several years The Peregrine Fund, Inc. has worked cooperatively with the Zoological Society of San Diego, and other zoos, to breed this eagle in captivity, with the goal to restore breeding pairs in suitable habitats throughout Central America (Burnham 1998).

The first exploratory hacking began in the Soberania National Park, not far from Panama City, in 1998, in cooperation with INRENARE, the Panamanian federal authority in charge of wildlife, and the Panama Audubon Society. Five eaglets were hacked at different times during the year from a 14-foot tower erected in a small glade within the forest. Since a young Harpy may remain with its parents for one to two years before becoming independent, we expected that continual monitoring and food-provisioning would be necessary for a long time. The eagles are tracked daily by radio telemetry. Surprisingly, the birds have already been observed to make kills on their own, including adult three-toed sloths and howler monkeys. Unfortunately, a poacher shot one of the birds, an all too common cause of death throughout the range of this majestic species, and another disappeared under suspicious circumstances (J. P. Jenny, W. Heinrich, pers.comm. 1998).

Ospreys

Roy Dennis has pointed out to me that interest in translocation of the Osprey *Pandion haliaetus* apparently antedates the contemporary spate of raptor projects by several decades. The British falconer and exhibitionist, Capt. C.W.R. Knight, obtained four nestlings from Gardener's Island, New York in August of 1929, transported them home, and released them in September as fully fledged juveniles on an islet in Loch Arkiag, Inverness-shire, Scotland, where the species reportedly had last nested in 1908 (Knight 1932, R. Dennis *in litt.* 1998.). The young Ospreys were artificially provisioned with food until they learned to fish for themselves, but nothing more is known about the fate of these birds (Table 3).

More recently Larry Rymon, East Stroudsburg University of Pennsylvania, undertook a reintroduction of the Osprey in five western counties of Pennsylvania (Schaddt & Rymon 1981, Rymon 1989). He hacked 111 wild-taken Ospreys from 1980 through 1985, and a further 125 birds from 1990 through 1995, in a region where no nesting Ospreys were known in 1981 (Houghton & Rymon 1997). Released birds began nesting five years after the first hack, and by 1998 there were approximately 35 pairs breeding widely across Pennsylvania, many of them released birds but also wild birds that were attracted into the population by the released ones; other hacked birds had dispersed into adjacent states (Houghton & Rymon 1997, L. Rymon *in litt.* 1998). (Table 1.)

Additional, extensive hacking of Ospreys in West Virginia, Kentucky, and Tennessee has allowed these birds to move into previously unoccupied range and to take advantage of new habitat created by artificial reservoirs and waterways (Westall 1990). "Overall, the Osprey population in these states surged from five nesting pairs in 1981 to 105 in 1994" (Houghton & Rymon 1997). The project in

Tennessee, for example, hacked 165 Ospreys from 1980 through 1989 (Hammer & Hatcher 1983, B. Hatcher *in litt* 1998); by 1998 there were 97 to 98 territorial pairs in the state, and 91 breeding pairs produced 171 young, most of them nesting on man-made structures (B. Hatcher *in litt* 1998). Again, this breeding population consists of birds hacked both in and outside Tennessee, as well as some naturally occurring wild birds that have been attracted to settle in the region. (Table 1.)

In 1983, Larry Gillette and Judy Voigt-Englund of Hennepin Parks began a project to translocate Ospreys by hacking in suburban areas of the Twin Cities; they were joined later by the Raptor Center at the University of Minnesota (Martell *et al.* 1994, Martell 1995). The project has released 155 young Ospreys in five different locations (M. Martell, pers.comm 1998). In recent years there have been 12 to 15 nesting pairs in the greater metropolitan area of St. Paul and Minneapolis, and others scattered more widely through Minnesota. As elsewhere, some of the breeders in this population are wild birds that appear to have been attracted to the area by the released ones. (Table 1.)

At the Rutland Water Nature Reserve in central England, a project is underway to translocate Scottish Ospreys into vacant range, under the auspices of the Highland Foundation for Wildlife, Leicestershire and Rutland Trust for Nature Conservation, and Anglian Water, plc. The project officer is Helen Dixon. In the first two years (1996-97) of a five year effort, 14 Ospreys have been hacked at a c.>1,000 hectare lake; 12 migrated, and one was reported dead in Senegal in February, 1997. Pairing is not expected until the third summer (Roy Dennis *in litt* 1998). (Table 2.)

Old World Vultures

In Europe and Israel there are many projects underway to reestablish populations of four species of Aegypine vultures, all threatened in one degree or another. The first, best reported, and most notable of these projects has been the successful effort in France to reestablish a breeding colony of the Griffon *Gyps fulvus* in the Grands Causses area south of the Massif Central from 1981 through 1986, spearheaded by the Fonds d'Intervention pour les Rapaces (Terrasse 1980, 1990, Terrasse *et al.* 1994, Sarrazin *et al.* 1994, Sarrazin *et al.* 1996). The release of 61 vultures over a period of six years eventually led to a population of 71 pairs in 1997 (F. Sarrazin *in litt* 1998). The main reasons for this outcome are (1) release of adult or near adult vultures from acclimatisation cages, resulting in early establishment of breeding pairs within one year after release, (2) high survival rates of both young (0.858) and adult (0.987) vultures, (3) subsequent breeding success of released and wild-produced offspring equal to that of wild birds in other areas (0.82 yng/pair), (4) high degree of philopatry (85% of wild-produced offspring settled in their natal gorges, and (5) conspecific attraction of dispersive individuals from other populations into the colony. The one drawback has been the necessity to support the colony with artificially provisioned food (mostly sheep carcasses), although the vultures have started foraging more on natural sources of carrion in recent years (Terrasse *et al.* 1994). (Table 1.) The success of this project in the Grands Causses area has led to several other Griffon translocations in France and in the Alps.

The international programme to restore the Bearded Vulture *Gypaetus barbatus* in the Alps (Austria, Germany, Italy, France), started in 1978, is also well publicized and has involved the cooperative effort of several zoos where the birds are bred for release, as well as various conservation groups, eg. Zoological Society of Frankfurt and World Wildlife Fund-Austria, all coordinated by the Foundation for the Conservation of the Bearded Vulture (Bijleveld 1979, Frey & Bijleveld 1984, Annual Reports, Foundation for the Conservation of the Bearded Vulture, 1994-96, Frey & Llopis 1998, H. Frey *in litt* 1998). (Table 2.) Bustamente (1996) has developed PVA models for both the captive and the translocated populations, emphasizing the need for more demographic data on the released birds.

In a similar way, the Black Vulture Conservation Foundation, started in 1986 and now headquartered in Majorca, Spain under the direction of Evelyn Tewes, coordinates several translocation projects for the European Black or Cinereous Vulture *Aegypius monachus* (Terrasse 1989, Tewes 1994, Tewes *et al.* 1998, E. Tewes *in litt* 1998). (Table 2.)

Translocation projects for the Egyptian Vulture *Neophron percnopterus* are also underway or planned in France and Italy.

Condors

1. Andean Condor *Vultur gryphus*, Tables 2 and 3—In the New World, captive breeding and reintroduction programmes have been established for both the Andean Condor and the California Condor. At first the Andean Condor was used experimentally as a "surrogate species" to develop techniques of translocation to be used for the critically endangered California species (Wallace & Temple 1987, Wallace 1991). In one project, 11 juvenile and immature, captive-bred condors were hacked, and one wild adult hard-released with back-pack transmitters on the Sechura Peninsula, northern Peru. The adult died after two months, but seven of the 11 captive-bred condors survived for more than two years when monitoring stopped. These birds had integrated socially with the local wild condors, but it is not known whether any of them became breeders. In 1980-1983 a release of 12 female Andean Condors was undertaken in southern California to test the current environment for future release of California Condors and to gain experience in the release of condors; the birds were later trapped and returned to captivity (Wallace & Temple 1987).

Because Andean Condors breed readily in captivity, a number of zoos keep breeding pairs, and several zoos in the American Zoological Association, under the leadership of the Zoological Society of San Diego, banded together to provide captive-bred condors for translocation projects in South America. The first of these projects started in the trans-Andean region of Colombia (four hack sites) in 1990, a cooperative venture of the Colombian Ministry of the Environment, the San Diego Zoological Society, the Cali Zoo, and The Peregrine Fund, Inc. (Lieberman *et al.* 1990, 1993, Lieberman 1994). A total of 42 condors has been released through 1998, and after nine years three breeding pairs are known to be established, one involving a wild mate (A. Lieberman *in litt.* 1998). Other projects are starting up, such as the one in Venezuela and one north of San Carlos de Bariloche, Argentina, on the eastern slope of the Andes where condors are rare, carried out in cooperation with the Buenos Aires Zoo (M. Wallace *in litt.* 1998).

2. The California Condor *Gymnogyps californianus*, Table 2—One of the few critically endangered species for which it was decided to remove all individuals from the wild for breeding in captivity, the last wild California Condor was trapped in 1987. At that time the entire population consisted of 27 condors, 10 captured as wild birds, four taken as nestlings, and 13 hatched from eggs taken in the wild (Toone & Wallace 1994, Kiff *in press*). The wisdom of this highly controversial decision can be seen 10 years later with the production of more than 110 young condors in captivity (1988-1998).

The California Condor Recovery Plan calls for the establishment of three separate populations of 150 birds each, two in the wild and one in captivity, but the plan has not been followed strictly. The release of captive-bred condors began in southern California in 1992, a cooperative effort of the U. S. Fish and Wildlife Service, San Diego Zoological Society, and Los Angeles Zoological Society. A total of 39 birds has been released over six years (through 1997) at three different release sites; eleven had to be returned to captivity owing to behavioural problems (perching on electric transmission structures, interacting with human beings at campgrounds, etc.). Of the remaining 28, nine have died as of 1998, but some of the older birds are beginning to show signs of pairing.

A second translocation project started in northern Arizona near the Grand Canyon in 1996, a cooperative effort of the Arizona Department of Fish and Game, U. S. Fish and Wildlife Service, U. S. Bureau of Land Management, and The Peregrine Fund, Inc. Nineteen birds have been hacked through 1997: Two have been killed, one disappeared (presumed dead), and one had to be trapped and removed to captivity because it was accepting food from human beings. Fifteen remained alive as of August 1998 and range widely over northeastern Arizona and southern Utah but still return to the hack-site on the Vermilion Cliffs for food.

A third project began in the Ventana Wilderness of the Los Padres National Forest, Monterey County, California in January of 1997, under the management of the Ventana Wilderness Sanctuary Research and Education Center). The first four condors had to be trapped and taken back because of too many behavioural interactions with human beings. A second group of five condors released in December of 1997 has fared better so far.

It is too soon to be certain, but it appears likely that California Condors will be breeding again in the wild before long. See Lloyd Kiff's paper in this volume for more details of this work.

Hawks and other Accipitrids

1. The Goshawk *Accipiter gentilis*, Table 1.—Always a rather rare bird in Great Britain, at least in recent centuries (Bannerman 1958), the Goshawk had more or less disappeared there by the end of the 19th Century. Breeding became regular again in the mid-1960s, and the species increased significantly in the 1970s and 1980s (Marquiss & Newton 1982), so that by the 1990s the population could be estimated as c. 400 territorial pairs (Petty 1994), with major centres around the Welsh-English border, the Peak District of Derbyshire/Yorkshire, and especially the English/Scottish border. In other areas, gamekeepers have kept the number of Goshawks at a low level (I. Newton, *in litt.* 1998). This population expansion resulted from the establishment of captive-held birds. Although 25 Goshawks of Fennoscandian origin were released experimentally by hacking or by hard-release of older birds, most of the introduced Goshawks were trained birds that escaped from falconers or were deliberately let loose after a period in captivity (Kenward *et al.* 1981, Marquiss & Newton 1982). It was estimated that an average of 20 imported Goshawks per year successfully entered the wild in the decade of the 1970s, when falconers were able to bring many birds into the country. Others have no doubt become established since then.

2. The Red Kite *Milvus milvus*, Tables 1 and 2—A highly organized effort to increase the distribution and numbers of the Red Kite in Britain has been underway since 1989, sponsored by the Joint Nature Conservation Committee, English Nature, Royal Society for the Protection of Birds, Scottish Natural Heritage, and the Forestry Commission of England (Ian Carter *in litt.* 1998, Roy Dennis *in litt.* 1998, Ian Evans *in litt.* 1998). Four locations were chosen in Scotland and England where the species had long been extirpated by human persecution but where the habitat appears to be highly suitable. Wild young for hacking have been obtained from Spain, Germany, Sweden, and a few from Wales. In southern England, workers hacked 93 kites between 1989 and 1994: Breeding occurred in the second year, and by 1997 there were >51 territorial pairs. Similarly, in northern Scotland, 93 kites were hacked between 1989 and 1993: Breeding first occurred in the third year, and in 1997 23 pairs were on territory. A project started in central England in 1995 has hacked 49 kites in three seasons, and four breeding pairs were found in 1997. The fourth project started in east central Scotland in 1996 has released 37 kites in two years, and two territories were occupied in 1997. The stunning success of this programme can be attributed to (a) optimum, vacant habitat, (b) high survival of the released kites from the first year on, (c) early breeding at 1 to 2 years of age, (d) a high reproductive rate, and (e) intelligent and dedicated management (Evans & Pienkowski 1991, McGrady *et al.* 1994, Carter *et al.* 1995, Evans *et al.* 1997, Evans *et al.* in press).

3. Harris' Hawk *Parabuteo unicinctus*, Table 1—The Santa Cruz Predatory Bird Research Group, in collaboration with the U. S. Bureau of Land Management, U. S. Fish and Wildlife Service, California Department of Fish and Game, and other agencies, undertook a reintroduction of the Harris' Hawk along a 320 km riparian corridor of the lower Colorado River from 1979 through 1990 (Stewart 1981, Walton *et al.* 1988a, B. Walton *in litt.* 1998). The project was part of a planned ecological restoration programme to re-vegetate a badly disturbed riparian zone with native shrubs and trees (Rosenberg *et al.* 1991), but unfortunately the plant restoration was never completed, leaving limited suitable habitat for the hawks, especially nesting sites. The Predatory Bird Group released 220 hawks by hacking, fostering, and immediate hard-release of adults including already paired birds. Some of the adults bred in the area within the first year of release; young birds began breeding in their second year. According to Brian Walton (*in litt.* 1998), the Harris' Hawk proved to be especially easy to translocate; however, because of the limited habitat, most of the released birds dispersed out of the riparian zone as soon as the few suitable nesting locations became occupied. Even so, three to five pairs of hawks have now bred in the area for 16 years, and a much larger population would no doubt have become established if the planned re-vegetation had taken place (B. Walton *in litt.* 1998). [It should be noted that Beck *et al.* (1994) incorrectly included the Harris Hawk among the 16 successful reintroductions out of the 145 projects they analyzed.]

4. Common Buzzard *Buteo buteo*, Table 3—An experimental study of buzzards translocated into Norfolk and Sussex, England, where wild buzzards are very sparse, has been undertaken by the Institute of Terrestrial Ecology, in collaboration with the Natural Environment Research Council,

National Trust, Nature England, and the British Falconers' Club, Robert E. Kenward, principal investigator (Kenward *et al.* 1996). A total of 44 radio-tagged birds were released by hacking from 1994 through 1998. A high dispersal from the release area has reduced the opportunity for local pairing, but even so five pairs were found on territory in 1998 (R. E. Kenward *in litt.* 1998), and continued releases would no doubt build up a viable population in this part of England.

5. Montagu's Harrier *Circus pygargus*, Table 1—At the Parc Natural dels Aiguànnolls del' Empordà in northeastern Catalonia, the Wildlife Service of Catalonia State has undertaken to restock Montagu's Harrier in a region where the species has been greatly diminished, using both captive-produced and wild-taken young for hacking (Pomarol & Parellada 1989, Pomarol 1994). Since 1988, 198 birds have been hacked, but few have returned to the area despite the high rate of survival to independence. Although hacking has been effective in releasing independent, capable juveniles into the wild, Montagu's Harriers have poorly developed philopatry, and most juveniles disperse widely across the landscape and do not settle near their natal localities to breed. Moreover, the habitat for harriers in Catalonia has been highly degraded by agriculture. Interestingly, since 1990 between two and six pairs of harriers have nested near the release site each year, but most of them are not released birds. Do they represent another instance of conspecific attraction? In such cases, a better technique might be to establish paired harriers in flight cages located in the desired habitat and then release them after their chicks have hatched. Site tenacity might then hold them to the area in future years, but harriers are noted for moving around from one area of high prey density to another in different years. Hacking is now being used to rescue wild nestlings in agricultural lands where nests are likely to be destroyed by ploughing or haying operations (M. Pomarol *in litt.* 1998).

Summary of Results

In summary, of the 28 well executed and documented projects (Table 1), I consider 21 (75%) to have succeeded in establishing or substantially aiding in establishing self-sustaining populations. Only three (10.7%) appear to have been failures, all owing to adverse environmental influences and not to inherent technological problems. The ultimate success or failure of four other projects cannot be clearly predicted at this time. Of the 15 on-going projects (Table 2), 10 (66.6%) appear highly likely to succeed, only two (13.3%) show a low likelihood, and four are too recent to judge. One of the experimental projects (Table 3), Common Buzzard in Britain, although designed for basic biological studies, appears likely to lead to significant augmentation of breeding numbers in a depopulated region. Work with the Harpy Eagle looks promising so far and could lead to successful reintroductions in Central America, where the species is virtually extirpated.

DISCUSSION

Successfulness of Raptor Translocations

In a detailed, stepwise regression analysis of variables associated with 134 translocations of birds and 64 of mammals, Griffith *et al.* (1989) identified eight factors associated with "success" (creation of a self-sustaining population). Their most significant findings were: (a) projects involving threatened species were less successful than those involving native game species (44% versus 86%), (b) releases in excellent habitat were more successful than those in poor habitat (84% versus 38%), (c) releases in the core of the historical range were more successful than those at the periphery or outside (76% versus 46%), (d) release of wild-caught animals was more successful than release of captive-reared ones (75% versus 38%), and (e) herbivores were more frequently established than carnivores (77% versus 48%). A re-analysis of their data further emphasized the difference in success between using captive-reared and wild-caught animals and also suggested that captive-reared birds are less often successfully translocated than captive-reared mammals (32% versus 62%) (Griffith *et al.* 1990). They offered the opinion that a greater than 50 per cent chance of success should usually be the minimum criterion for starting a translocation project.

A follow-up study (Wolf *et al.* 1996), also using multivariate analysis, mostly confirmed the original findings after six additional years, except that the earlier difference between captive-reared

and wild animals was reduced to marginal significance ($P = 0.034$), although for birds as a subset the difference remained 71% versus 40%. Beck *et al.* (1994) also reviewed 145 reintroductions of captive-produced animals, and reported only 16 cases (11.0%) of successfully established populations in nature, based on a minimum establishment of 500 self-sustaining individuals or a PVA demonstrating self-sustainability.

Contrary to some of the figures and analyses summarized above, my evaluation of raptor translocations indicates a high rate of successful establishments—at least 75 per cent of those projects that have been carried out with sufficient scope (Table 1), a figure that could grow to 80 per cent or more in time. This degree of success compares to a reported 63–67 per cent for birds in general, 72–73 per cent for mammals in general, 48 per cent for “carnivorous” birds, 94 per cent for carnivorous mammals, 48 per cent for herbivorous birds, 67 per cent for herbivorous mammals, 81 per cent for omnivorous birds, and 67 per cent for omnivorous mammals (Wolf *et al.* 1996). Griffith *et al.* (1989) also found that success rates approaching 70 per cent were associated with the release of large numbers of threatened and endangered birds in habitats of high quality; 68 per cent of the translocations in Table 1 involved the multiple release of >100 individual birds in good to high quality habitat.

Considerable caution needs to be exercised, however, in comparing estimates of success among different studies and even among compared projects in the same study, because different assumptions and perceptions of population characteristics often underlie an individual researcher’s determination of success or failure (Sarrazin & Barbault 1996). In the absence of a PVA or other probability analysis based on real data (Bustamente 1996), such determinations are best made on the basis of objective criteria like those adopted by the IUCN Species Survival Commission for its “Red List categories” (Mace & Lande 1991, Cade *et al.* 1997). For example, the evaluations by Griffith *et al.* (1989, 1990) and Wolf *et al.* (1996) give the appearance of rigour and scientific objectivity because they employ complicated statistical analyses yielding all kinds of coefficients, chi-square values, and P -values; but in the final analysis, all these statistical results are based on the independent and subjective evaluations of *ca.* 200 respondents who decided whether or not their projects had resulted in “self-sustaining populations,” the criteria for which were not specified.

Furthermore, there is the problem of scope in defining individual translocation projects and its influence on project outcome. How meaningful is it to compare a single translocation of x number of individuals at one location with multiple translocations at several sites? In their analysis, Griffith *et al.* (1990) included 18 Peregrine Falcon projects and 15 Bald Eagle projects among a total of 39 “carnivorous bird” translocations. I was surprised to learn that there were 18 Peregrine projects in North America! What they must have done was to obtain state agency reports on hack-site projects within their respective borders and considered each state as a separate project, whereas I have always considered the activity within states to be part of a regional programme. One can reach quite different results from these two perspectives. If I were to judge each of the 15 eastern USA states where Peregrines have been released as separate projects uninfluenced by the others, I would probably find two complete failures, at least five that had not achieved a self-sustaining population, and no more than eight that were successful, but when they are viewed as integral parts of one overall regional programme they represent a highly successful reintroduction (Cade *et al.* 1997).

The scope of a project also has a temporal dimension that must be taken into account. A project may fail to achieve a self-sustaining population in its first 10 years but will do so in the following five to 10 years. The period of translocation for the 21 successful projects in Table 1 averaged 13 years, ranging from 5 to 20 years, and Beck *et al.* (1994) reported an average of 11.8 years for the projects they identified as successful. Griffith *et al.* (1989) also found that length of effort, as well as number of released animals, was highly correlated with successful translocation. Thus, there is little heuristic value in comparing recent, on-going, or short term projects with these longer running efforts.

Wilson and Stanley Price (1994) noted that raptor translocations are among the more successful avian projects. There are both intrinsic, species-specific factors and extrinsic, environmental factors that probably account for the relative ease with which raptors can be successfully translocated. I can only list them here as hypotheses for further study. Intrinsic factors: (1) Raptors, especially the

larger ones, are less subject to predation than many other kinds of birds; (2) they appear to be physically stronger than other birds of comparable size; (3) they are often easier to keep in captivity with less physical deterioration; (4) they appear to be more resistant to diseases than many other birds; (5) many species show strong philopatry, making re-population of a specific area by translocation easier; (6) raptors tend to be K-selected species with high rates of juvenile and adult survival, so that a high percentage of translocated individuals remain in the population and survive to breed; even though delayed sexual maturity and low rates of productivity are associated with the high survival, once established a breeding pair can be expected to produce young over periods of 5 to >10 years, depending on the species; (7) raptors show conspecific attraction (Stamps 1988, Smith & Peacock 1990, Reed & Dobson 1993), and individuals can find each other over vast distances, so that a small translocated population has the capability of recruiting wild individuals into its ranks (both conspecific and heterospecific attraction occur, especially among colonial nesting birds, McIlhenny 1934, Cade 1986, Cade & Temple 1995); and (8) raptors have a high degree of behavioural acceptance of, and adjustment to, hands-on manipulations in the field, and they can often be acclimatized to new or modified environmental conditions, thereby expanding their distributions and increasing their abundance.

Extrinsic factors: In addition to those relating to the natural environment (see Griffith *et al.* 1989, 1990; Wolf *et al.* 1996), I would like to call attention to two that are especially relevant to the birds of prey. First, the methods employed in the sport of falconry, particularly hacking, have contributed a ready-made, centuries-old and tested technology that, with little modification, has been adapted to raptor translocation with a high degree of success. When I contemplate the large number of hacking projects and the thousands of hack-site attendants of the past 25 years, I sometimes wonder what the Emperor Frederick II of Hohenstaufen, were he to be re-enthroned amongst us, would think of all this activity occurring 700 years after he established a scientific basis for the practice of falconry in his famous treatise, *De Arte venandi cum Avibus*.

Finally, most of the people who become involved in raptor translocation tend to have “green thumbs” as Dillon Ripley (1975) aptly noted: People “who have an innate skill which probably could never be learned and certainly has nothing to do with the possession of a higher educational degree.” They become fanatically devoted to their birds, so that they willingly exert the extra effort and insight needed for success. “A sense of kinship with nature and a single-mindedness of purpose appear to be the touchstones of success in this work” (Ripley 1975). Statisticians have not yet learned how to factor this human dimension into multiple regression analysis.

Ethical and Societal Concerns

These topics range from concerns about animal welfare and humane treatment of captive held and translocated animals, and the degree of naturalness that can be expected from such manipulations, to worry about the need for greater participation by professional scientists and use of scientific method in translocation projects. All of these concerns have been repeatedly addressed by one official body or another beginning with the 1976 “Manifesto on Animal Reintroductions” produced by the British Section of the World Wildlife Fund, continuing to the 1987 “IUCN Position Statement on Translocation of Living Organisms,” to the most recent “Re-introduction Guidelines” (1998) of the IUCN Species Survival Commission’s Re-introduction Specialist Group (see Kleinman *et al.* 1994); but discussion of these issues continues unabated. For instance, a round table discussion on bird reintroductions at the International Ornithological Congress at Durban, South Africa in August of 1998 focused all its attention on “Improving the rigour of re-introduction project assessment, planning, and execution,” starting from the premise that in the past many projects have been characterized by poor planning, absence of theoretical scientific underpinnings, and lack of involvement by professional biologists (from summary provided by the Chairman of the Bird Section, IUCN SSC Re-introduction Specialist Group, Philip Seddon, *in litt.* 1998). In the following paragraphs I consider some of these concerns as they relate to raptor translocations.

Ethics and moralistic sentiments about humane treatment aside (Polivitis 1990), it should go without saying that animals reared or held in captivity for the purpose of translocation must be

maintained in good health and in fit condition both physically and psycho-behaviourally for survival and eventual reproduction in the wild after release (Cade 1980). It is hard to imagine that managers of a legitimate translocation project would consider releasing physically or behaviourally handicapped animals. Raptor translocations enjoy a great advantage in this regard, because the techniques for handling and maintaining raptors in fit condition in captivity have been perfected over a period of 3,000 years in the practice of falconry, and many new innovations in husbandry and health care have also been added in recent years (Redig 1993, Fox 1995, Heidenrich 1997). There is no excuse based in lack of technology for maintaining captive or manipulated raptors in poor condition.

As to the naturalness of raptors re-established by translocation, generally speaking they appear to be virtually indistinguishable from their wild counterparts in behaviour, demography, and ecology, particularly after one or two generations beyond the released founders (e.g., Green *et al.* 1996, Sarrazin *et al.* 1996, Tordoff & Redig 1997). On the other hand, some captive-produced or hand-reared individuals of certain species do show behavioural modifications that can be either disadvantageous or advantageous for survival and reproduction after release. Perhaps the commonest of these manifestations is increased tameness or tolerance of close human activities. This has been a problem for released California Condors, because these big, conspicuous birds are extremely inquisitive and exploratory, especially in their early years of life, and they often investigate human activities ranging from remote fishermen's camps to tourist centres in national parks. Released Peregrines and Mauritius Kestrels have shown remarkable plasticity in their settlement behaviour for breeding, and in addition to choosing natural sites for nesting, many of them historically known eyries, they also accept a wide variety of new, often man-made sites in drastically altered habitats (Cade & Jones 1993, Jones *et al.* 1995, Cade *et al.* 1996a). The same is true to a lesser extent for Bald Eagles and Ospreys. Such behaviour has allowed these species to extend their breeding distributions into areas formerly considered to be unsuitable. Also, in a few instances it has been necessary to provide supplemental food to reintroduced raptors—particularly the carrion-feeders [e.g., Griffon Vulture in France (Terrasse *et al.* 1994), and California Condor in California and Arizona (Kiff *in press*)], although artificial provisioning should not be required indefinitely.

Population geneticists, and people persuaded by them, worry incessantly about what happens to genotypes and gene pools during the process of captive breeding and translocation (see Templeton 1990 for a balanced view). Snyder *et al.* (1996) stressed seven negative aspects of using captive-bred animals for translocation, especially the influence of "domestication" (loss of genetic fitness for existence in the wild). Loss of fitness for the wild from unnatural selection in captivity usually occurs in higher vertebrates only after many generations, and the way to avoid this problem is to design the breeding programme so that all birds needed for release are produced in the first few generations in captivity. To my knowledge no diurnal raptor translocation involving captive-bred birds has gone beyond the F_3 to F_4 generations. The more serious concern, actually, is the potential for phenotypic modification or loss of essential behaviour patterns in captivity, especially for species that rely heavily on social transmission of "traditions" from one generation to the next, as Snyder *et al.* (1996) correctly point out. In other matters of concern to geneticists, inbreeding depression, if it has occurred, has not been a problem for the Mauritius Kestrel, which was reduced to only two surviving pairs at one time in its history but now numbers several hundred wild-ranging individuals; nor has the cross-breeding of half a dozen subspecies of Peregrine Falcons for release in the eastern third of the United States (Temple & Cade 1988) been associated with any noticeable outbreeding depression. Some of these "hybridized" Peregrines are among the most superb specimens to be seen anywhere in the world, and so are their numerous offspring.

Adverse impacts on biotic communities have also been concerns, especially the potential spread of diseases from translocated animals to existing wild populations (Woodford & Rossiter 1994, Snyder *et al.* 1996, Cunningham 1996). Although diseases have infrequently ravaged captive populations of raptors, there is no evidence so far that any translocation of raptors has been associated with the spread of disease to wild animals. It remains a *potential* problem, which the precautions set forth by Cunningham (1996) and the Re-introduction Specialist Group's guidelines adequately address, while the stricter regimen advocated by Snyder *et al.* (1996) is unnecessarily restrictive and

costly. Common sense precautions such as quarantine prior to release and observation of physical and behavioural signs of poor condition are sufficient in most cases. Problems associated with predation by translocated raptors on domestic livestock and poultry, and on endangered species of prey, sometimes occur locally and have to be dealt with by negotiations among the concerned parties on a case by case basis (Cade *et al.* 1996b). Competition with other predatory species is also a problem, but I am unaware that a case has arisen so far.

Translocation and associated "single species" programmes are often criticized for competing with "better techniques" such as habitat preservation and ecosystem management for the scarce money allocated to conservation (Myers 1979, Pitelka 1981, Csuti *et al.* 1987, Hutto *et al.* 1987, Varner & Monroe 1990, Snyder *et al.* 1996). This criticism presumes that there is an objective basis for judging which methods are better than others. It also presumes that if particular allocations of money were not spent on translocation they would automatically go to support the better methods. Neither assumption has credibility. Much of the money that supports captive breeding and translocations comes from sources that otherwise would not be funding conservation at all (Conway 1995). Government agencies do have to prioritize the use of their appropriations for conservation, and tough decisions often have to be made; consequently some constituencies will not be satisfied by how the money is spent.

Conservationists should spend less time coveting one another's sources of income and more time convincing the rest of humankind to use a greater portion of its wealth on conservation. The world economy is measured in the many trillions of dollars, much of it wasted on frivolous and unnecessary, if not dysgenic, activities. The 100's of billions of dollars spent annually on drug addictions alone would more than support all needed conservation efforts (see Cade 1988b).

A related fear has to do with the danger that some people will view hands-on methods such as captive breeding and translocation as convenient, substitute strategies for habitat preservation and maintenance of natural ecosystems (Snyder *et al.* 1996). The fact that politicians and some bureaucrats, and some corporate executives, find this argument expedient is not a fault of the methodology or of its practitioners; it is the fault of those who attempt to beguile the public with such notions. For example, the mere suggestion that captive breeding and translocation could be valuable aids in supporting the distribution and abundance of the Spotted Owl *Strix occidentalis* was met with irrational resistance by the entrenched conservation establishment associated with the owl from fear that the forest industry would latch onto the idea as an excuse to forgo needed forest habitat protection (Jack Ward Thomas, *in litt.* 1990). Again, it is up to the conservation community to keep the record clear, without denigrating a proven and valuable methodology.

Lastly, there is the repeated plea that scientists and the scientific method should be more involved in translocation projects (Scott & Carpenter 1987, Nisbet 1988, Griffith *et al.* 1989, 1990, Kleinman *et al.* 1994). Sarrazin and Barbault (1996) decry the lack of cooperation between scientific ecologists and conservation managers in the field and make the point that reintroduction (translocation) projects are an important way to do field experiments relevant to theoretical ecology (e.g., Sarrazin *et al.* 1996, R. Kenward, unpublished data on the Common Buzzard). They expressed disappointment that Beck *et al.* (1994) found "professional biologists" involved in only a little more than half of the 145 reintroduction projects they analyzed; what Beck *et al.* (1994) actually reported was professional training of graduate students in biology associated with 56 per cent of all projects. That percentage is higher than one might have expected. Among the 52 raptor projects listed in Tables 1-3, no fewer than 32 (61.5%) enjoyed the close involvement of professional biologists at the Ph.D. level, and all but four (92%) were managed by professionals trained in some aspect of applied biology such as wildlife management.

While I agree that translocation projects offer unique opportunities to investigate certain scientific questions relating to demography, behavioural ecology, and other matters and that the application of biological theory and principles to management is essential for optimum success (Cade 1974, 1980, 1988b, Cade *et al.* 1996b), I am less certain that academic scientists necessarily make the best managers or should be the ones in charge of translocation projects, unless they are done solely for research purposes. I rather suspect that most should not be, because pure scientists tend to be mainly

interested in their theories and in testing hypotheses, not in practical results. A good translocation manager is someone who combines common sense and hands-on abilities with a practical concept of how to make a project work.

Scientists, on the other hand, are undoubtedly the best people to interpret the results of a translocation and to predict outcome from an analysis of relevant data. Scientific input can help in adjusting the tactics of translocation to fit the demographic and behavioural characteristics of particular species. For example, what is the optimum pattern of release for K-selected species versus r-selected species: is it better to release a large number of individuals simultaneously for the latter, while spreading the releases out over a longer period for the former, or does it matter one way or the other? How does one go about re-establishing a highly dispersive species that has poorly developed philopatry, e.g. Montagu's Harrier? Whenever possible scientists and managers should work together on translocation projects, but only a few academic or pure scientists have shown serious interest, despite the unique opportunities available to them (Sarrazin & Barbault 1996).

CONCLUSION

In 1984, the role of translocation in raptor conservation was just beginning to be defined when I offered some speculations on its future (Cade 1986). Work in the intervening years has largely confirmed the view I outlined then: Techniques such as captive breeding and translocation should be used as methods of last resort when it becomes clear that a species or population cannot survive in the wild without such intervention. Given that caveat, translocation can be used in several ways to promote conservation. (1) It can be used to restore populations to vacated range, or, in some cases, to extend the range of a species into new areas. (2) It can be used to increase the size of small populations, especially by augmenting reproductive output by the addition of nestlings or juvenile birds. (3) It can be used to maintain the presence of a species in patches of habitat (e.g. parks and reserves) that are too small to support a self-sustaining population through time. (4) It can be used to replenish the genetic composition of small populations that suffer from inbreeding depression or other genetic problems, by the addition of individuals with different genotypes, and (5) it can be used to modify behaviours and ecological adjustments that allow individuals to survive and breed in degraded or changed habitats, thereby increasing overall species distribution and abundance (Cade 1986).

In short, translocation has proved to be effective in restoring rare and threatened raptor populations in nature, and its importance will no doubt increase as more and more birds of prey become threatened by habitat degradation, fragmentation, and loss. Like other forms of hands-on management, it lies at one end of a spectrum of actions needed to conserve biological diversity, with ecosystem management at the other end (Cade & Temple 1995). It is no better than other methods of conservation, but it also is no worse. Although the more holistic actions focused on ecosystems and habitats are essential to maintain maximum diversity, to prevent additional species from becoming vulnerable to extinction, and to provide ecological scope for the recovery of endangered species, once a species becomes threatened, it usually requires special attention focused on its particular needs for restoration to a safe population size and demographic composition. Translocation can often meet those needs more effectively than natural processes.

On the other hand, the general impression that one is likely to gain from reading all the discussions and commentaries in the literature of the past 20 years is that translocation and related hands-on methods are largely failures, and a waste of time and money, if not actually harmful in their consequences. This is a false impression as far as birds of prey are concerned, and I believe it is likely false in a more general sense as well. Because of this derogatory attitude prevalent in some quarters of the scientific and conservation communities, the practitioners of translocation have indulged themselves in an inordinate amount of introspection, self-criticism, and rule-making (see, e.g., letters on pp. 3-5 in *Conservation Biology*, 11(1), 1997). I suggest it is now time to accept that there will be some failures, to continue learning as much as we can from our efforts, take justifiable satisfaction in our considerable successes, and move on to the work ahead of us.

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