7

Populations: re-introductions

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7.1 Re-introduction of species and restoration ecology

According to the Society for Ecological Restoration (SER International; SER 2002), ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed. The very first 'attribute of restored ecosystems' mentioned in the *SER Primer on Ecological Restoration* (SER 2002; www.ser.org) is that 'The restored ecosystem contains a characteristic assemblage of the species that occur in the reference ecosystem and that provide appropriate community structure.' In this context, intended introductions of species are an important tool, because dispersal is very often a major constraint, especially in highly fragmented habitats and landscapes. Thus, restoring diversity is a crucial part of ecological restoration, but while the SER (2002) considered it primarily in an ecosystem context, the issue of the re-introduction of species has also frequently been considered at the species, or subspecies, level. For example, Falk *et al.* (1996), in their volume on strategies for the re-introduction of endangered plant species, considered re-introductions also as a **conservation** tool. In this chapter I will examine experiences with re-introductions, independent of whether they have been performed in a strict restoration context or rather as a species-conservation tool. Indeed, re-introductions are nearly always experiments and the science of re-introduction is in its infancy, which urges us to learn from earlier experiences (Falk *et al.* 1996).

In response to the increasing occurrence of re-introduction projects worldwide and to help ensure that the re-introductions achieve their intended conservation benefit, the Re-introduction Specialist Group of the International Union for the Conservation of Nature's (IUCN's) Species Survival Commission has developed guidelines (IUCN 1995), which are implemented in the context of the IUCN's broader policies pertaining to biodiversity conservation and sustainable management of natural resources. According to the IUCN, the principal aim of any re-introduction should be to establish a viable, free-ranging population in the wild, of a species, subspecies or race, which has become globally or locally extinct, or extirpated, in the wild. The population should be re-introduced within the species' former natural habitat and range and should require minimal long-term management. The objectives of a re-introduction may include the enhancement of the long-term survival of a species, the re-establishment of a keystone species in an ecosystem (or an emblematic species from a cultural point of view), or the maintenance and/or restoration of biodiversity in (semi-)natural landscapes. In the literature, both the terms re-introduction and translocation are being used. Strictly speaking these terms do not mean exactly the same thing. A re-introduction is an attempt to establish a species in an area that was once part of its historical range, but from which it has been extirpated or become extinct. Re-establishment is often used as a synonym, but implies that the re-introduction has been successful. A translocation is a deliberate and mediated movement of wild individuals or populations from one part of their range to another.

Community, ecosystem and landscape changes and transformations carried out in the past may have great consequences for the success of re-introduction attempts because the ecosystem or habitat at issue may have become permanently unsuitable for the species of concern. However, even when habitat suitability is ensured, many re-introductions nevertheless fail. In some cases it has been hypothesized (Law & Morton 1996, Lundberg *et al.* 2000) this could be caused by 'community closure'; that is, the feasible and persistent community to which the lost species once belonged is no longer 'open' for reinvasion. This approach could, for example, help explain the results of a study on dispersing prairie voles (*Microtus ochrogaster*) by Danielson and Gaines (1987). In this experiment, voles were introduced into enclosed resident populations of the same species, of southern bog lemmings (*Synaptomys cooperi*), of cotton rats (*Sigmodon hispidus*) or into an empty enclosure. The results indicated that colonization by dispersing voles was negatively affected most by resident conspecifics. Introduced female voles were more strongly affected than males during the growing season but not during the non-growing season when reproductive activity was typically low. Resident bog lemmings also negatively affected colonization by dispersing voles, but after the colonization phase coexistence was possible. Cotton rats did not affect colonization by dispersing voles. Further investigation is required to reveal to what extent this kind of interspecific interaction within guilds plays a role in re-introduction attempts.

There are, however, many other factors involved in re-introduction becoming either a success or a failure. Wolf *et al.* (1996) evaluated 80 translocations of birds and mammals in Australia, New Zealand and North America, and compared the results with a similar analysis carried out in 1987 by Griffith *et al.* (1989). The analysis revealed that approximately 58% of all translocations conducted with thousands of individuals of threatened, endangered or sensitive birds and mammals have failed to establish self-sustaining populations. Furthermore, keystone species play a critical role in communities, and their effects are generally much larger than would be predicted from their relative abundance. The importance of keystone species is essentially recognized through removal experiments (Paine 1966; see also Chapter 2 in this volume). A keystone species often referred to is the sea otter (*Enhydra lutris*), living in the north Pacific. Sea otters feed on sea urchins (*Strongylocentrotus franciscanus*), which in turn feed predominantly on kelp (macroalgae; e.g. Mate 1972). If keystone species become threatened or go extinct in their habitat it can be expected that the system changes dramatically and that, next to trying to re-introduce the keystone species into its habitat, the changes may have become so large that re-introduction becomes very difficult. Recently it has been shown through modelling work that even (random) removal of species can lead to cascading extinctions far beyond the target one (Borvall *et al.* 2000, Lundberg *et al.* 2000), and that cascading extinctions are positively related to species abundance and connectance (Law 1999). If extinctions are followed by community closure, re-introductions are even more difficult. If ordinary (i.e. non-keystone) species can already have such effects, what can we expect if keystone species become extinct? No clear field data are available at present, but this question stresses the need for the conservation of keystone species while they are still present in their original habitat.

In the remainder of this chapter I will highlight some of the important aspects of the art and science of re-introducing species that largely determine either success or failure.

7.2 Source populations

Individuals to be re-introduced can come from various sources and as a first step a careful assessment should always be made of the taxonomic status of the candidate sources or provenances. Even though the species concept as a basic taxon unit is controversial, individuals should ideally be of the same subspecies as those that were locally extirpated. Genetic studies should be carried out, if possible, to determine the relative degree of taxonomic and genetic similarity between possible substitutes and the pre-existing population. Genetic analyses may also permit prediction of the likelihood of hybridization taking place with other taxa in the target or release area or region. For animals, it is preferable that source animals derive from wild populations. For plants and animals, the source population should ideally be closely related genetically to the original native stock and also show ecological characteristics (morphology, physiology, behaviour, habitat preference, etc.) similar to the original population or subpopulation.

If a subspecies has become extinct in the wild and in captivity, a substitute form may be chosen for possible release. Such substitutions are actually a form of benign introduction. Selection of a suitable substitute should focus on extant subspecies and consider genetic relatedness, phenotype, ecological compatibility and the conservation value of potential candidates. For example, a local population of ibex (*Capra ibex*) that became extinct in Czechoslovakia was replaced by re-introductions of Austrian *C. ibex* and Turkish *Capra hircus aegagrus* and *C. ibex nubiana* from the Sinai desert (reviewed in Stanley-Price 1989). The inevitable hybrid forms dropped their kids in the middle of the winter, 3 months earlier than pure *C. ibex*, resulting in the death of all offspring. This case illustrates the need to assess both hybridization risks and ecological compatibility (Seddon & Soorae 1999). In general, there is a need for information on whether the introduction can literally be considered a re-introduction or whether it entails a risk of effects like those related to unintended invasions by aliens (see Chapter 2 in this volume).

Removal of individuals for re-introduction should not endanger the wild source population, and individuals should only be removed from a wild population after the effects of translocation on the donor population have been assessed and evaluated. When removals from source populations are large relative to its size, problems may arise (Stevens & Goodson 1993). Sometimes a species may become so threatened in the wild that it is taken into captivity, and the loss of wild animals may leave only captive populations. Examples include the Arabian oryx (*Oryx leucoryx*), the Przewalski horse (*Equus przewalski*) and the Sorocco dove (*Zenaida graysoni*; Stanley-Price 1989). In such cases there is still the potential to breed species in captivity although the results of genetic and phenotypic changes such as genetic drift, inbreeding, domestication, increased tameness and the loss of behavioural traits will tend to preclude the chances for successful re-introduction and subsequent *in situ* conservation. However, many attempts are and should be made to conserve and restore critically threatened species through the re-introduction of captive-bred animals into suitable habitats. Recent examples include programmes for the black-footed ferret (*Mustela nigripes*), the golden lion tamarin (*Leontopithecus rosalia*) and the red wolf (*Canis rufus*). Unfortunately, the success rate of re-introduced captive-bred individuals is highly variable and often very low (James *et al.* 1983).

A special hazard to successful re-introductions of animals is the risk of disease introduction. The guidelines of IUCN (1995) prescribe that prospective release stock should be subjected to a thorough veterinary screening process before transport from the original source. There are many examples of devastating effects of diseases introduced unintentionally. From 1893 to 1906, 332 elk (*Cervus canadensis*) were released in the Adirondack region of New York. Additional animals were released in 1916 and 1932. The releases initially appeared successful, and in 1906 the population was estimated at 350 elk. However, the elk slowly disappeared, and there has not been an authenticated report of elk in the Adirondacks since 1953. The parasitic round worm *Pneumostrongylus tenuis* was the likely cause of the failure of elk to survive in the Adirondacks (Severinghaus & Darrow 1976). However, it is not sure when the round worm appeared for the first time in this area.

7.3 Founding numbers, diversity and population structure

In general, the number of individuals that are released in re-introduction attempts is small. This means that founding groups are susceptible to the same dangers of increased extinction risks as small, natural populations: environmental fluctuations, demographic stochasticity and inbreeding. Therefore, to achieve the highest possible success, a primary goal of re-introduction should be to maximize the initial rate of population increase in order to shorten the period during which the introduced population is exposed to these risks. This can be brought about by releasing a high number of individuals in a high-quality habitat. Komers and Curman (2000) investigated how the rate of increase of more than 30 newly re-introduced populations was affected by various population characteristics such as population size, sex and age structure in Artiodactyla (even-toed ungulates). Their results were in line with the general notion that re-introduction success increases with the number of animals released (Fig. 7.1). The function became asymptotic at about 20 animals. When fewer than 20 animals were released, the variance in growth rate increased substantially and, of a number of factors, only the age structure explained a significant portion of this variance. The population growth increased

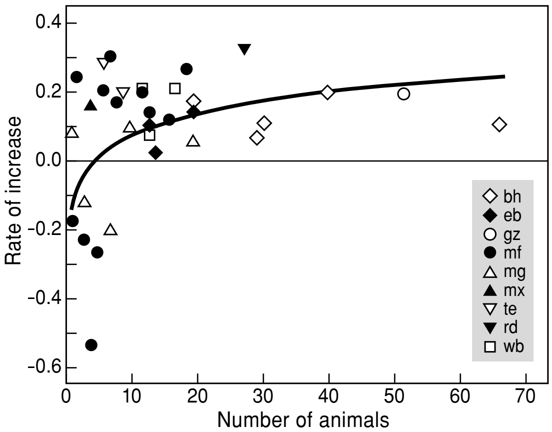

Fig. 7.1 The rate of population increase in relation to the number of animals in re-introduced ungulate populations. Source species are shown in the key: mf, mouflon; mx, muskox; te, tule elk; rd, red deer; mg, mountain goat; eb, European bison, wb, wood bison; bh, bighorn sheep; gz, mountain gazelle. After Komers and Curman (2000).

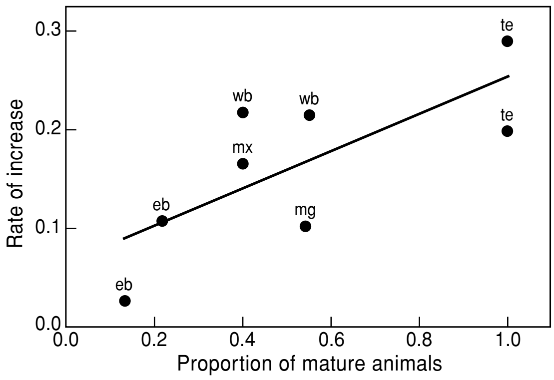

Fig. 7.2 The rate of population increase in re-introduced ungulate populations in relation to the proportion of socially mature animals in populations of fewer than 20 animals. Source species: mx, muskox; te, tule elk; mg, mountain goat; eb, European bison; wb, wood bison. After Komers and Curman (2000).

with the proportion of mature animals in the population (Fig. 7.2). This finding could be explained by a higher fecundity of mature females.

Loss of genetic variability, due to genetic drift and/or inbreeding, is especially likely when an effectively small number of individuals is used in founder populations. Because of its importance, many conservation plans call for the maintenance of genetic variability in translocated populations. Stockwell *et al.* (1996) studied the effects of translocation in mosquito fish (*Gambusia affinis* and *Gambusia holbrooki*). These fish have two life-history traits that should minimize the loss of genetic variability; they have high reproductive potential, and females retain sperm and commonly have multiply sired broods, maximizing the ratio of the effective population size N_e to the total population size N (see Chapter 6). Ten translocated populations were examined. These populations had significantly lower levels of heterozygosity than their respective parental source populations. The most striking result was a reduction in allelic diversity in the translocated populations that varied from 24 to 40%. All losses were of relatively rare alleles, and probably due to an undocumented bottleneck in the early introduction history. The results were surprising because initial translocations involved hundreds of fish and because mosquito fish, as mentioned above, have various reproductive traits that appear to minimize the effects of bottlenecks (on genetic diversity). Similar effects have been found in other introduced populations (e.g. seabream, trout, salmon, Anolis lizards, house sparrow, common myna, reindeer and ibex), as reviewed by Stockwell *et al.* (1996). In 50% of the cases examined, translocated populations had lower heterozygosity than their parental sources. In approximately 75% of the cases, refuge populations had reduced levels of allelic diversity. This pattern agrees with theoretical expectations: founding events should have a stronger effect on allelic diversity than on heterozygosity (Nei *et al.* 1975, Allendorf & Leary 1986). Also, reductions in allelic diversity are often due to loss of rare alleles, which typically have little effect on overall heterozygosity. It is clear that re-introduction programmes should attempt to create populations with high levels of genetic diversity. However it will not be easy to prevent some loss of genetic diversity. Starting with the highest possible number and ensuring a high initial population growth rate will help to maintain high genetic diversity.

Sometimes, insights in metapopulation theory may be used to understand the success rate of translocations.

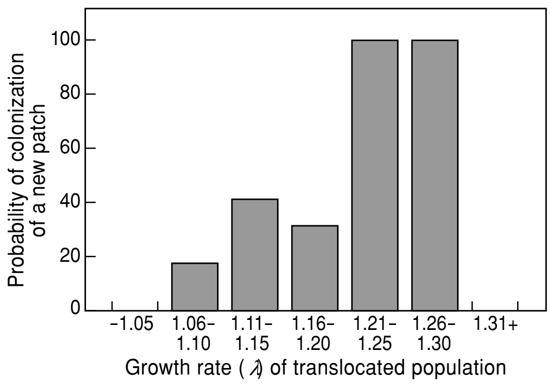

Fig. 7.3 Probability of successful colonizations of new patches is correlated with population growth rates (λ) of 31 translocated populations of bighorn sheep in the western USA in 1946–97. After Singer *et al.* (2000).

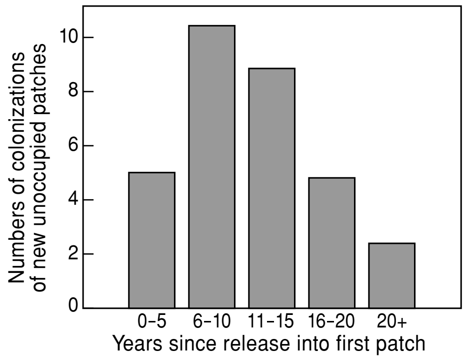

Fig. 7.4 Probability of successful colonizations in relation to the number of years since release in the first patch for 31 translocated populations of bighorn sheep released 1947–91. After Singer *et al.* (2000).

Singer *et al.* (2000) related the fate of a number of bighorn sheep re-introductions to this species naturally occurring in metapopulations. At present, most extant populations of bighorn sheep (*Ovis canadensis*) consist of feweer than 100 individuals occurring in a fragmented distribution across the landscape, whereas the species formerly occupied a more continuous and wider range. They investigated the correlates for the rate of colonization of 79 suitable, but unoccupied, patches by 31 translocated populations of bighorn sheep released into nearby patches of habitat. Dispersal rates were 100% higher in rams than in ewes. Successful colonizations of unoccupied patches (24 out of 79 patches were colonized) were associated with rapid growth rates of the released population (Fig. 7.3), years since release (Fig. 7.4), larger area of suitable habitat in the release patch, larger population sizes and a seasonal migration tendency in the released population (Fig. 7.5). In this study area, colonization rates were much higher than other studies have reported and this could be attributed to the presence of larger regions of unoccupied suitable habitat with a greater probability for detection than the other studies. It is possible that bighorn sheep existed mostly in metapopulations but that human disturbance has accelerated extinction rates in these metapopulations, and that bighorn sheep now occur in a non-equilibrium state. The results of this study also indicate that many restoration projects in the past probably suffered from poor procedures. Many prior translocations consisted of small founder groups (typically fewer than 25 animals) released into small, isolated patches of habitat, probably representing a near-perfect prescription for failure.

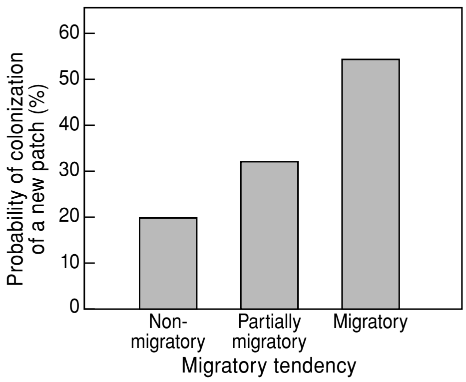

Fig. 7.5 Probability of successful colonizations in relation to migratory tendency in the release patch for 31 translocated populations of bighorn sheep. Migratory, > 75% of the population uses distinct seasonal changes; partially migratory, part of the population migrates; non-migratory, year-round use of the same ranges. After Singer *et al.* (2000).

7.4 The re-introduction site

Re-introduction in the core of the historic range is sometimes indicated to be better than along the periphery (Griffith *et al.* 1989, Wolf *et al.* 1996). However, Lomolino and Channell (1995, 1998) found that 23 out of 31 species of endangered mammals persisted along the periphery, not in the core or central portion of their historic range. In addition, persistence was greater for insular than for continental populations. According to Lomolino and Channell, the range periphery, in comparison with core sites, encompasses a much more diverse collection of habitats and environmental conditions. They referred to the California condor (*Gymnogyps californianus*) as an important case in point. Which range should investigators adopt as the raptor's historic range? Recent efforts include release to a site in northern Arizona, well outside the condor's present range, but also providing protection from anthropogenic threats. From their review of re-introductions of Marsupialia in Australia, Short *et al.* (1992) came to the conclusion that the success rate of island (re-)introductions (60%) was far greater than those in mainland Australia (11%), even though the successful island (re-)introductions were all to islands with no historic record of the occurrence of the (re-)introduced species. Success of (re-)introduction of these macropods appeared to depend critically on control or exclusion of exotic terrestrial predators such as foxes and cats. Peripheral sites should thus not automatically be discarded as suitable re-introduction sites.

A crucial aspect of any re-introduction plan is an assessment of the availability and quality of the re-introduction site. Re-introductions can have a chance of success only if the habitat and landscape requirements of the species are or could be satisfied, and are likely to be sustainable. The area should have sufficient carrying capacity to sustain growth of the re-introduced population and support a viable self-sustaining population over time. Identification and elimination, or reduction to a sufficient level, of previous causes of population decline and/or habitat transformation should take top priority. In a habitat suitability study for an otter re-introduction project in Utah, it was found that 94% of the studied streams were unacceptable for re-introductions. Escape cover was the most limited habitat attribute, whereas food for otters appeared to be available in adequate quantities (Bich 1988). This study, therefore, recommended that no otter re-introductions should be made until riparian zones were rehabilitated and protected, since re-establishment of stream-bank vegetation was deemed essential to provide escape cover for re-introduced otters. Similarly, Howells and Edwards-Jones (1997) studied the feasibility of re-introducing wild boar (*Sus scrofa*) to Scotland through an assessment of suitable woodland habitat that could support a minimum viable population of the target animal. This species has been the focus of early attempts to re-introduce it into Britain. Based on a review of *S. scrofa* ecology, the authors identified woodland habitats suitable for supporting wild boar. Only long-established woodlands containing some stands of semi-natural origin and larger than 500 ha in size were considered. None of the woodlands could be considered optimum habitat for wild boar and none was large enough to support a minimum viable population of 300 animals. The study concluded that the goal of (re-)establishing a self-sustaining population of wild boar in Scotland was unrealistic in the short term.

Habitat destruction and modification can also be brought about in the form of invasive species. Such invasions often result in dramatic changes in ecosystem structure or function (Gordon 1998, Hobbs & Mooney 1998). Invasive species may not only lead to changes in ecosystem properties but can also hamper re-introductions through predation. Recent attempts to recover razorback suckers (*Xyrauchen texanus*), an endangered piscivorous fish species, by re-introducing them into their native range of mainstream Colorado River have not been successful because of predation on the native young suckers by non-native fishes (Johnson *et al.* 1993). In another study, Bergerud and Mercer (1989) reviewed 33 (re-)introductions of caribou that took place in eastern North America between 1924 and 1985. Twenty introductions resulted in sustained populations and 13 failed, the majority as a result of predation by wolves. The fate of these 33 introductions is consistent with the view that predation (natural and hunting) is a major factor in the decline of caribou in eastern North America following European settlement. In Europe, meanwhile, attempts to re-introduce black grouse (*Lyrurus tetrix*) and capercaillie (*Tetrao urogallus*) have been hampered, in part by predation by goshawk (*Accipiter gentilis*) and pine marten (*Martes martes*; Kalchreuter & Wagner 1982). Predation may have particularly severe impacts on very small populations, especially if a more common primary prey species is present (prey switching), while at the same time the number of re-introduced individuals is almost always small or very small. Only a sufficiently large re-introduction might overcome predation and succeed where a smaller one would fail. The minimum viable population would then, however, be much larger than that predicted by standard population-viability analysis. When McCallum *et al.* (1995) used a simple stochastic model based upon the bridled nailtail wallaby (*Onychogalea fraenata*) to explore this possibility, even very small amounts of predation (two to four individuals per 6 months) were sufficient to cause large re-introductions of up to 50 animals to fail. No clear threshold population size was found beyond which re-introductions would succeed. The moral is that if predation is a serious risk, a single re-introduction of a given size is preferable to multiple re-introductions of the same total number of individuals.

7.5 Re-introduction techniques

In the majority of cases of animal re-introductions or translocations, the focus is on populations, rather than communities, whereas for plants the focus is on communities. Many techniques are applied to help new animal populations to get established. Use can be made of individuals caught in the wild or of individuals kept and bred in captivity. Choices have to be made concerning which stages in the life cycle of species are most suitable for translocation activities. Should it be eggs/seeds, subadults/seedlings, or adults? For plants, individual plants or whole turfs can be transplanted, or seed mixtures can be harvested from hay and spread over the introduction site. A few commonly applied techniques will be discussed below.

7.5.1 Plants

Community translocation involves the wholesale removal of an assemblage of species from a site and the attempt to establish it as a functioning community at a receptor site. The translocation of species assemblages is used to move communities that would otherwise be completely destroyed by a change in land use at the donor site (e.g. civil engineering and excavation projects; Klötzli 1987). Bullock (1998) reviewed, among several others, 10 translocation projects in Britain. Four different techniques were used: hand turfing, machine turfing, macroturfing (1 m × 2 m) and spreading (of excavated soil and vegetation). In most projects, post-translocation management was similar to the original management at the donor site. All communities, except the species-poor heath, showed both losses and gains of species. At some sites, all translocated communities were becoming more similar to the original communities at the receptor sites. Rare plant species were lost on a regular basis. The associated invertebrate communities showed larger and more obvious changes than did the plant communities, and often showed losses in rare species of conservation importance.

The restoration of a former plant community *in situ* is quite another issue. If seeds of the target species (characteristic of the original plant community) are no longer available in the soil seed bank, they have to immigrate from elsewhere, for example attached to haymaking machinery or after deliberate re-introduction. Somerford Mead is an old flood-meadow along the River Thames near Oxford, UK, which harboured a *Alopecurus pratensis*/*Sanguisorba officinalis* plant community in the 1950s. From 1960 to 1982, however, it was used as grassland for haymaking or silage cutting and received artificial fertilizers. From 1982 to 1985 it was ploughed and used for barley. In 1985 it was agreed to take Somerford Mead out of this high productivity and set in motion regimes to create an *Alopecurus*/*Sanguisorba* flood-meadow community again. The possible benefits of removing the surface soil to reduce fertility was set against the disadvantage of synchronously removing much of the seed bank. Therefore, in 1986 the last crop of barley was grown without any fertilizer in order to start the reduction of nutrient availability. Further restoration efforts have been described by McDonald (1992, 1993, 2001) and McDonald *et al.* (1996). In July 1986 a seed mixture was harvested from the reference site Oxey Mead, an ancient flood meadow, 2 km downstream. Its exploitation has not changed since at least the 13th century (Baker 1937). It features, therefore, a notably low-fertility grassland community, the *A. pratensis*/*S. officinalis* association (MG4, according to Rodwell 1992). The seed mixture was broadcast over prepared soil on Somerford Mead the following October. Management included cutting for hay in early July and grazing the aftermath by cows and/or sheep. During the first 3 years target species such as *Bromus* spp., *Cynosurus cristatus*, *Festuca pratensis*, *Leucanthemum vulgare*, *Ranunculus* spp., *Rhinanthus minor* and *Trisetum flavescens* had become established. After 6 years, 20 target species were found in the established vegetation that had not been re-introduced from the reference site. They must have spread spontaneously or by haymaking machinery. The position of many re-introduced target species became critical. *Silaum silaus* and *Leontodon hispidus* occurred in the seed bank and only rarely in the established vegetation. *A. pratensis*, *Briza media*, *Hordeum secalinum* and *S. officinalis* were not found in the seed bank and were rare in the established vegetation. Species with short-lived seeds cannot form a seed bank, and hence cannot survive years when they are absent from the established vegetation.

In 1989 a management experiment began in Somerford Mead consisting of an annual hay cut at the end of June followed by 4 weeks of grazing in October – by sheep or cattle – in comparison to a control, non-grazed treatment. From 1990 onwards, the differences between grazed and ungrazed treatments increased. The ungrazed plots became dominated by tall grasses such as *Arrhenatherum elatius*, *Dactylis glomerata*, *Festuca rubra*, *Holcus lanatus* and *Lolium perenne*. At the same time, the frequency of *Bromus hordeaceus*, *Cirsium arvense*, *C. cristatus*, *Ranunculus bulbosus*, *Trifolium pratense* and *Trifolium repens* decreased. The ungrazed plots changed in composition towards *Arrhenatherum elatius* grasslands common on road verges in Britain. Both the cattle- and sheep-grazed plots became more similar to the community in the reference site, but were still far from the species composition of Oxey Mead, even 15 years after the re-introduction of target species.

7.5.2 Fish and herpetofauna

The restoration of historical spawning areas, or the provision of new, suitable spawning habitat, are important for successful re-introduction of fish and amphibians. Both translocation from the wild and the release of captive-bred individuals are commonly applied techniques. Stocking appropriate life stages of target species is clearly important for successful introductions or re-introductions. For fish, using older/larger individuals has been more successful than using spawn (Noakes & Curry 1995), whereas the reverse seems to have been the case for amphibians. Cooke and Oldham (1995) monitored the establishment of large populations of common frogs (*Rana temporaria*) and common toads (*Bufo bufo*) for 6 years in a newly created reserve, following stocking with spawn of both species and with toads rescued from a site to be destroyed. Transfer of spawn proved to be more effective as a means of establishing a new population of toads than transfer of adults.

The Great Lakes ecosystem has changed dramatically in the past 50 years. A review of historical changes reveals complex interactions of overexploitation of fishery resources, invasion of non-indigenous species, eutrophication, extensive habitat modification and toxic contamination. Native fish species that required tributary or near-shore habitat for spawning and nursery areas have declined markedly. Among surviving native species, such as walleye (*Stizostedion vitreum*), stock diversity declined with the loss of tributary-spawning stocks and lake-spawning stocks became dominant. With the rarefaction of native species, the abundance of formerly subdominant species increased. Species such as smelt (*Osmerus mordax*), gizzard shad (*Dorosoma cepedianum*) and white perch (*Morone americana*) depend less on critical tributary and near-shore habitat (Koonce *et al.* 1996). Invasive species pose a special problem. The Great Lakes ecosystem is home to at least 139 non-indigenous species of fauna and flora that have become established following invasions or intentional introductions. About 10% of the exotic species have caused economic or ecological damage to the system. Despite activities to reduce the causes of decline, most problems have not yet been solved adequately. Nevertheless, several re-introduction attempts have been made with various species. Much attention has been given to the rehabilitation of the lake trout (*Salvelinus namaycush*). It seems that a complete restoration of the Great Lakes is unlikely, due to naturalization of exotic species, habitat degradation and destruction, heavy fishing mortality, lack of native gene pools and complicated political jurisdictions that rarely work towards a common vision. Meffe (1995) proposes that a more realistic goal would be rehabilitation, a movement along the trajectory towards complete restoration.

Until now, most re-introduction projects involving amphibians and reptiles have not been very successful (Dodd & Seigel 1991), but efforts undertaken for the natterjack toad (*Bufo calamita*) represent an interesting exception. The species is endangered in Britain and has been legally protected since 1975. This amphibian suffered a major decline during the first half of the 20th century, due partly to habitat destruction but mostly to successional changes in its specialized biotopes and anthropogenic acidification of breeding sites. Extensive autecological research over the past 25 years has provided the foundations for an intensive, 3-year species-recovery programme funded by statutory nature-conservation organizations. Management of heath and dune habitats focused on restoration and maintenance of early stages of serial succession, initially through physical clearance of invasive scrub and woodland vegetation, followed by applying grazing regimes similar to those prevalent in earlier centuries. In some cases extra breeding pools were constructed to either increase or stabilize natterjack toad populations that had become reliant on one or very few pools at small sites, or to promote range expansion within large habitat areas. Re-introductions also had been attempted. At least six out of 20 re-introductions resulted in the foundation of expanding new populations, and an additional eight have shown initial signs of success. Conservation methods developed for *B. calamita* provided a useful precedent for long-term conservation of early successional habitats and species (Denton *et al.* 1997).

7.5.3 Birds

In birds, making use of captive-produced eggs that are fostered or cross-fostered is a common and viable re-introduction technique (Derrickson & Carpenter 1983). Sometimes eggs are collected from wild populations. Fostering has proved to be a much better technique than the release of hand-reared individuals as they are much more prone to all sorts of danger (e.g. predation). This has been found, among others, in whooping cranes (*Grus americana*), hand-reared capercaillie (*T. urogallus*), white storks (*Ciconia ciconia*) and raven (*Corvus corax*). Releasing individuals straight into the wild (hard release) is not recommended by Bright and Morris (1994). Most species of birds (and mammals) rely heavily on individual experience and learning as juveniles for their survival. They should be given the opportunity to acquire the necessary information to enable survival in the wild. Therefore, soft-release techniques have been developed whereby the animals are kept in pens or other holding devices and slowly are made acquainted with their new environment.

A commonly used soft-release technique for the introduction of birds of prey is called hacking. Hacking is the release of free-flying young birds at a site where food is provided until independence. Hacking was used in the re-introduction of Montagu's harrier (*Circus pygargus*; Pomarol 1994). It took place in an enclosure measuring 3–4 m × 2 m × 1 m high. The re-introduced harriers were between 20 and 30 days of age. After 5–8 days the enclosure was opened. The young birds became independent on average 34 days after their first flight (at 70 days of age). Over a 5-year period 87 birds were (re-)introduced with a success rate of 83%. Only three birds had been seen returning to the area in subsequent years. Hacking has also been applied very successfully in the many re-introduction projects of the peregrine falcon (*Falco peregrinus*). Over the past 25 years more than 1000 birds have been re-introduced in this way in many parts of the USA.

More than 1670 attempts have been made to establish several hundred avian species worldwide. Among them are many raptors. At least six species of owls and 15 species of diurnal raptors have been established successfully. Examples of raptors that have been re-introduced or newly introduced are little owl (*Athene noctua*) in Britain, eagle owl (*Bubo bubo*) in Sweden and Germany, goshawk (*A. gentilis*) in Britain, white-tailed sea eagle (*Haliaeetus albicilla*) in Scotland and other parts of Europe, bald eagle (*Haliaeetus leucocephalus*) in New York and California, Seychelles kestrel (*Falco araea*) on Praslin (Seychelles) and the peregrine falcon (*F. peregrinus*) in the USA, Canada and Germany. A raptor that went almost extinct is the Mauritius kestrel (*Falco punctatus*). By 1974, the species had declined to only four known wild birds, including one breeding pair, as a result of habitat loss and pesticide contamination. A conservation project begun in 1973 has used many management techniques including captive breeding, supplemental feeding of wild birds, provision of nestboxes, multiple clutching, egg pulling, artificial incubation, hand rearing and release of captive-bred and captive-reared birds by hacking, fostering and predator control. A total of 331 kestrels were released in the 10 years up to the end of the 1993–4 breeding season; one-third of these were captive bred and the rest were derived from eggs harvested from the wild. By the 1993–4 season, an estimated 56–68 pairs had established territories in the wild with a postbreeding population, including floating birds and independent young, of 222–86. Since the pesticides responsible for their decline are no longer used, the number of Mauritius kestrels should continue to rise through natural recruitment. The distribution of suitable habitat suggests that an eventual population of 500–600 kestrels on Mauritius is possible. Due to its outstanding success, the release programme for the Mauritius kestrel ended after the 1993–4 breeding season (Jones *et al.* 1995).

7.5.4 *Mammals*

Mammals can be taken from wild source populations or from captive breeding stock. Catching animals from the wild can be a costly and time-consuming operation, and is not without risk. Like birds, also mammals should be given the opportunity to acquire the necessary information to enable survival in the wild, and soft releases are therefore recommended. Mammals propagated in an enclosure tend to develop an affinity for their immediate surroundings and therefore, upon release, exhibit a slow dispersal rate. This behaviour generally enhances survival. An example of a successful soft release is the case of the scimitar-horned oryx (*Oryx dammah*) in Tunesia which disappeared from that country in 1902 due to desertification, competition with domestic livestock, disturbance and hunting. Ten young scimitar-horned oryx (five males, five females) from Britain were re-introduced into the Bou-Hedma National Park in Tunisia in December 1985. They were acclimatized in a 600-m^2 pen for 4.5 months and then released into a 10-ha pre-release enclosure. Social organization was established peacefully, and the oryx adjusted to the new climate and natural foods. In July 1987, the oryx were released from the enclosure into the total-protection zone of the park. This zone is a 2400-ha area that has been protected from domestic livestock since 1977 (Bertram 1988).

Sometimes a more spectacular technique is indeed the only solution. In a translocation project for beaver (*Castor canadensis*) in Idaho, the mountains, heavy forests and lack of roads in Idaho made transplanting a labour-intensive, expensive and time-consuming task. In addition, it resulted in high beaver mortality. The use of planes and parachutes with animal holding boxes proved to be a much more efficient and much less expensive method of transportation. In 1948, 76 live beavers were dropped with only one casualty. Observations made in 1949 showed that the beavers that had participated in the airborne transplantation had settled and were well on their way to producing colonies (Heter 1950).

7.6 Socio-economic aspects and concerns

The California population of sea otter (*E. lutris*), in the 1970s introduced from Amchitka, Alaska, is considered vulnerable and therefore a Fish and Wildlife Service Recovery Plan for the sea otter has been made which calls for the establishment of a second California population as a hedge against devastation by a possible oil spill. During the 1970s transplant operations, 86 animals were captured, 24 of which died in the nets or holding pens. Of the 79 otters caught in 1971, 15 died due to capture complications (Mate 1972). Also the second translocation was not without its problems, including emotional reactions of various groups with widely different interests (Booth 1988). This example demonstrates that re-introductions can be highly controversial, especially with species that combine a high cuddling status with potential negative interaction with economic interests. The European habitat guideline 92/43/EEC demands a proper consultation of the public in case of re-introduction of species listed in Annex IV of the guidelines (EC 1992a).

Re-introductions are generally long-term projects that require the commitment of long-term financial and political support. It is important that socio-economic studies should be made to assess impacts, costs and benefits of the re-introduction programme to local people. According to the IUCN (1995) guidelines, a thorough assessment of attitudes of local people to the proposed project is necessary to ensure long-term protection of the re-introduced population, especially if the cause of a species' decline was due to human factors (e.g. over-hunting, over-collection or loss or alteration of habitat). The relevance of these guidelines should not be underrated because there are many examples of failures due to not paying sufficient attention to the attitudes of local communities. From an extensive literature review on exclosures, afforestation, reafforestation, rehabilitation and other regeneration operations over several million hectares in Mediterranean bioclimatic areas from the Atlantic Ocean to the Aral Sea, combined with 50 years of personal field experience, Le Houérou (2000) concluded that, while the main constraint for success is the restoration of habitat factors that have caused degradation, the most difficult constraints to overcome are usually of a socio-economic and/or sociocultural nature. Poaching can also be a problem, for example in the relocation of 22 tule elk (*Cervus elaphus nannodes*) from the Tupman Tule Elk Reserve near Buttonwillow to Fort Hunter Liggett (both in California) in 1978. Factors conducive to the high poaching rate were tameness of the relocated elk, location of release site, lack of monitoring and resentment by locals to changing policies at Fort Hunter Liggett (Hanson & Willison 1983).

Resentment can be especially strong against predators. Thus, when nine European lynx (*Lynx lynx*) were released in central Austria in 1975, 100 years after the last native lynx had been killed, there was strong local opposition from hunters, especially in Carynthia. Carynthia has few federal forest estates, but many large private forest estates pursuing trophy hunting by tourists as a source of income (Gossow & Honsig-Erlenburg 1986). Similar problems are encountered with wolves. In response to popular resistance, red wolves (*Canis rufus*) re-introduced to the Alligator River National Wildlife Refuge in North Carolina were classified as a 'non-essential experimental population' and did not have the full protection of the Endangered Species Act when released. Proposed re-introduction of grey wolves (*Canis lupus*) to Yellowstone National Park met similar opposition from livestock interests, hunters and state agencies (Wilcove 1987).

Some species have a much more positive press. Especially, the release of high-profile flagship species may raise public awareness of conservation issues and generate funding for wider programmes. In Saudi Arabia the first wildlife conservation project targeted the Houbara bustard (*Chlamydotis macqueenii*), which is threatened as a resident. Programmes directed towards the re-introduction of this large, appealing bird have attracted wide public attention owing to the emblematic status of the bird throughout the Middle East as the premier quarry for falconry, and thus these programmes have helped generate support for other, lower-profile species in need of protection. The aesthetic value or economic benefits of an animal may also be tied to the generation of public support and the means to raise public awareness of conservation issues. In Latvia the re-introduction of the beaver (*Castor fiber*) resulted in the creation and conservation of wetlands; their value in water purification has been estimated at up to £1.3 billion sterling, and beavers re-introduced into France and Sweden have become tourist attractions (Seddon & Soorae 1999).

In conclusion it seems to be clear that the idea of re-introducing species within their former habitat has gained quite some acceptance within the context of the restoration paradigm. An important incentive is that in most cases species are not able to colonize these areas by themselves and need a little help. Nevertheless, as has been amply demonstrated, much can go wrong and indeed has gone wrong in the many thousands of re-introduction attempts already set in motion. The ones that were successful, however, also teach us that it can be done and that success cannot be attributed to sheer luck alone. If re-introduction programmes take into account that the habitat is suitable (or can be made suitable again), the founding population is sufficiently large, the population structure is right, a high level of genetic diversity is ensured, the proper techniques are applied, careful planning has been applied and the public has been consulted properly, then the chances for a successful re-introduction are enhanced considerably.