1

# Animal Translocations: What Are They and Why Do We Do Them?

*Philip J. Seddon*[1], *W. Maartin Strauss*[2] *and John Innes*[3]

[1]Department of Zoology, University of Otago, New Zealand
[2]Department of Environmental Sciences, UNISA, South Africa
[3]Landcare Research, Hamilton, New Zealand



*'Translocation is now well entrenched as a conservation tool, with the numbers of animals being released in reintroduction and re-enforcement projects increasing almost exponentially each year.'*

Page 23

## Introduction

For as long as people have been moving from one place to another, which is as long as humans have been 'human', animals and plants have been moved with them, often hidden, unnoticed or ignored, but also as valued cargo. These so-called 'ethnotramps' include economically and culturally favoured species such as deer, macaque, civets, wallabies, cassowaries and wild-caught songbirds that were commonly carried around with humans (Heinsohn, 2001).

The variety of animals shown to have been translocated by prehistoric human colonists has been described as 'astonishing', with archaeological evidence of numerous and widespread human-mediated introductions as far back as tens of millennia, during the Pleistocene (Grayson, 2001). For example, it has been shown that people moved wild animals from the New Guinea mainland

*Reintroduction Biology: Integrating Science and Management*. First Edition.
Edited by John G. Ewen, Doug P. Armstrong, Kevin A. Parker and Philip J. Seddon.

to and between islands to the east and west over at least the past 20 000 years, for food and trade items as humans expanded their distribution and sought to retain access to animals whose habits were already known to them (White, 2004). It was during the Holocene (from ~11 000 years before the present), however, that the translocation of non-domesticated animals into novel habitats became one of the most significant human impacts on native animal populations (Kirch, 2005).

Clearly there are many reasons to translocate animals and some broad-scale classifications have been proposed, for example to distinguish between conservation translocations and those for commercial or amenity values (Hodder & Bullock, 1997), and along the way the terminology relating to translocations has become confused, contradictory and ambiguous. In this chapter we provide a framework for classifying the different motivations for animal translocation. We propose a simple decision tree that will enable conservation managers to categorize easily the different types of translocation, from reintroductions to assisted colonizations, and standardize the terminology applied in the species restoration literature. Throughout this chapter terms given in *italics* are defined in Box 1.1.

Box 1.1 **Glossary and definitions**

| | |
|---|---|
| *Analogue species* | Closely related form that could be used as an *ecological replacement* for an extinct species (Parker *et al.*, 2010) |
| *Assisted colonization* | *Translocation* of species beyond their natural range to protect them from human-induced threats, such as climate change (Ricciardi & Simberlof, 2009a) |
| *Assisted migration* | Synonym for *assisted colonization* |
| *Augmentation* | Synonym for *re-enforcement* |
| *Benign introduction* | Synonym for *conservation introduction* |
| *Biological control* | Intentional use of parasitoid, predator, pathogen, antagonist or competitor to suppress a pest population (Hoddle, 2004) |
| *Classical biocontrol* | The introduction of exotic natural enemies to control exotic pests (Thomas & Willis, 1998) |

| | |
|---|---|
| *Conservation introduction* | An attempt to *establish* a species, for the purposes of conservation, outside its recorded distribution but within an appropriate habitat and ecogeographical area (IUCN, 1998) |
| *Ecological replacement* | *Conservation introduction* of the most suitable extant form to fill the ecological niche left vacant by the extinction of a species (Seddon & Soorae, 1999) |
| *Ecological restoration* | The process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed (SER, 2004) |
| *Establishment* | Survival and successful breeding by founder individuals and their offspring; this is a prerequisite for, but not a guarantee of, population *persistence* |
| *Follow-up translocation* | Where one or more additional translocations are conducted to supplement an initial population established by *reintroduction* (Armstrong & Ewen, 2001) |
| *Introduction* | Intentional or accidental dispersal by a human agency of a living organism outside its historically known native range (IUCN, 1987) |
| *Managed relocation* | Synonym for *assisted colonization* |
| *Marooning* | *Translocation* to a predator-free offshore island |
| *Persistence* | The likelihood of population decline or extinction over some appropriate taxon-specific time frame |
| *Re-enforcement* | Addition of individuals to an existing population of conspecifics (IUCN, 1998) |
| *Re-establishment* | Synonym for *reintroduction* that implies the *reintroduction* has resulted in *establishment* (IUCN, 1998) |

| | |
|---|---|
| *Rehabilitation* | The managed process whereby a displaced, sick, injured or orphaned wild animal regains the health and skills it requires to function normally and live self-sufficiently (IWRC, 2009) |
| *Reintroduction* | Intentional movement of an organism into a part of its native range from which it has disappeared or become extirpated in historic times (IUCN, 1987) |
| *Reintroduction biology* | Research undertaken to improve the outcomes of *reintroductions* and other *translocations* (Armstrong & Seddon, 2008) |
| *Relocation* | Synonym for *translocation* |
| *Restocking* | Synonym for *re-enforcement* |
| *Restoration ecology* | The science upon which the practice of *ecological restoration* is based (SER, 2004) |
| *Species restoration* | The application of any of a wide range of management tools, including *translocation*, that aim to improve the conservation status of wild populations |
| *Subspecific substitution* | A subset of *ecological replacement* where the replacement taxon is a subspecies (Seddon & Soorae, 1999) |
| *Supplementation* | Synonym for *re-enforcement* |
| *Translocation* | Movement of living organisms from one area with free release in another (IUCN, 1987) |
| *Transplantation* | Synonym for *translocation* |

## The translocation spectrum

Seddon (2010) defined a conservation *translocation* spectrum, ranging from *reintroductions* through to forms of *conservation introduction*. Figure 1.1 broadens the scope and provides a framework for considering all motivations for moving wild animals. The first, simple, bifurcation divides movements into those that are accidental or incidental and those that are intentional

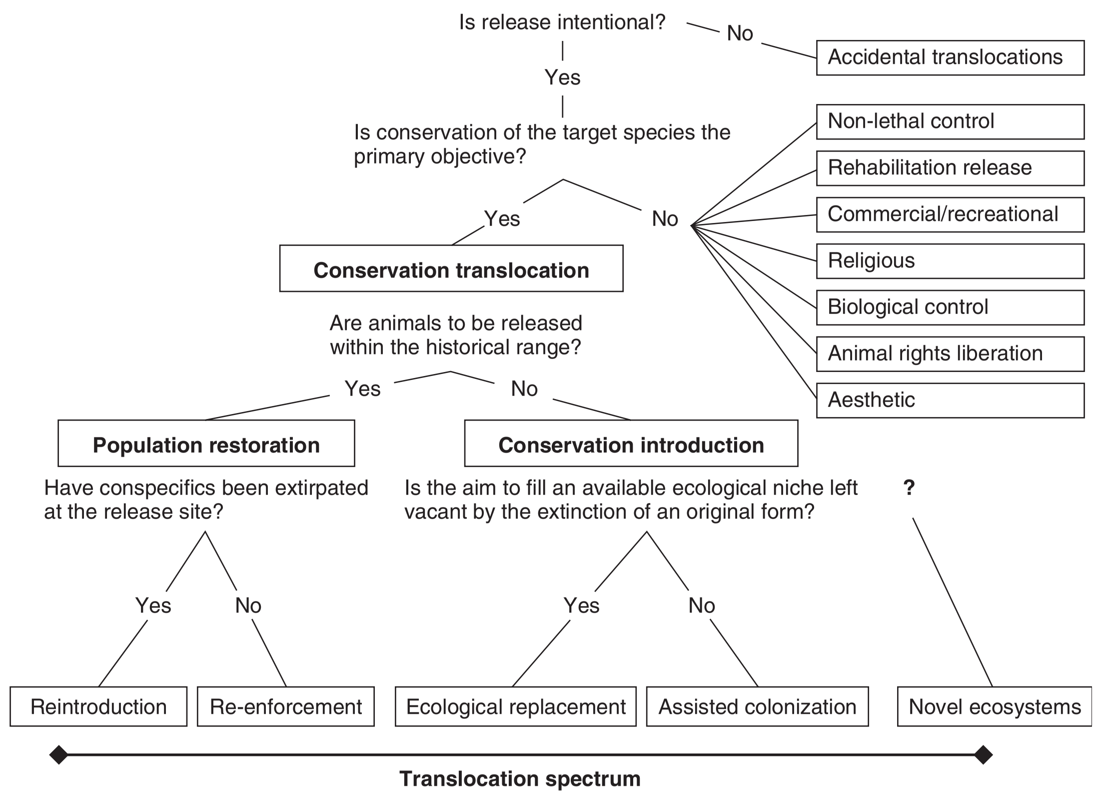


Figure 1.1 **The translocation spectrum.**

(Figure 1.1). Strictly speaking, accidental movements of wild animals are not translocations in the sense intended by the 1987 definition (IUCN, 1987). This IUCN definition, however, lacks mention of intent contained within a later, but confusing, redefinition (IUCN, 1998) that erroneously appears synonymous with re-enforcement and wild-to-wild movements. We take it that *translocations* are the deliberate and mediated (IUCN, 1998) movement of organisms, from any source, captive or wild, from one area to free release in another (IUCN, 1987). Thus *translocation* is the overarching term.

Not all translocations relate to the conservation of the species being moved. The next division in our framework therefore asks the question: is conservation of the target species the primary concern? (Figure 1.1). A split between conservation and non-conservation is in some senses simplistic and naive. Multiple, sometimes indirect, conservation benefits may accrue through translocations for, for example, recreational, commercial or wildlife rehabilitation motivations, not the least being opportunities for increased public engagement with nature and the enhanced public support for conservation measures that can arise from this engagement. Nevertheless, it is useful to make the distinction around primary concerns as many translocations may have multiple objectives and not uncommonly enhancement of the conservation status of the species may exist as a secondary goal.

## Non-conservation translocations

There are at least seven types of translocation for which conservation is not the primary aim (note that species conservation may be an associated aim and protection of individual animals of endangered species may be a primary aim): non-lethal management of problem animals, commercial and recreational, biological control, aesthetic, religious, wildlife rehabilitation and animal rights activism. One of the characteristics of many of the non-conservation translocations is that they are introductions, with the sometime exception of non-lethal management and wildlife rehabilitation.

### *Non-lethal management of problem animals*

As urban, suburban and agricultural landscapes spread, and where natural populations of wildlife species recover or expand, the potential for human–wildlife conflicts increases. In the United States, for instance, many states have Nuisance Wildlife Control Operator (NWCO) programmes in response to increased complaints of urban wildlife conflicts (O'Donnell & DeNicola, 2006). The most common forms of conflict are predation of livestock (Bradley *et al.*, 2005), attacks on humans and their domestic pets (Goodrich & Miquelle, 2005) and damage to property (Gammons *et al.*, 2009; Herr *et al.*, 2008). In large part due to public attitudes, lethal management of so-called problem wildlife is not a favoured option and instead a standard method of dealing with problem individuals is to capture and translocate them away from the focal point of conflict. The numbers of animals involved can be significant; for example, in 1994 in Illinois alone NWCO permittees moved >18 000 animals, including >13 000 raccoons (*Procyon lotor*), squirrels (*Sciurus* spp.) and bats, and throughout the US some hundreds of thousands of animals are moved to mitigate conflicts (Craven *et al.*, 1998). By far the majority of problem animals are mammals, with the larger species most commonly carnivores including ursids (*Ursus* spp.), felids (*Panthera* spp., *Felis concolor, Lynx lynx*), wolves (*Canis lupus*), and mustelids (*Mustela* spp.), but translocation of raptors, including the golden eagle (*Aquilia chrysaetos*), black eagle (*A. verreauxii*), crowned eagle (*Stephanoetus coronatus*) and martial eagle (*Polemaetus bellicosus*), has taken place in the United States and South Africa to reduce livestock predation (reviewed in Linnell *et al.*, 1997). Translocation of hen harriers (*Circus cyaneus*) has been considered to mitigate the impact of harrier predation on red grouse (*Lagopus lagopus scotica*) on UK moorland managed for grouse shooting (Watson & Thirgood, 2001). The priorities for problem animal translocations are primarily solving the specific conflict and secondarily the welfare of the individual animal. It is a public perception that translocated problem animals 'live happily ever after' (Craven *et al.*, 1998). In the few instances where post-release monitoring of such translocations has taken place, the reality is very different. Translocated problem animals typically show high post-release mortality due to the stress of capture, transport and release, aggression by territorial conspecifics, poaching, disorientation, unsuitable habitat, long-distance dispersal and disease (Craven *et al.*, 1998; Fischer & Lindenmayer, 2000). Carnivores in particular show strong homing behaviour (Bradley *et al.*, 2005), with many individuals able to return to the capture site and resume nuisance behaviour from release sites between 150 km (Lenain & Warrington, 2001) and up to nearly 500 km (Linnell *et al.*, 1997) away. Even when translocated animals do not return to the focal point of conflict, they tend to settle away from the release site and may become a new problem elsewhere (van Vuren *et al.*, 1997; Le Gouar *et al.*, this volume, Chapter 5). Consequently, translocation of nuisance animals is often ineffective at achieving either of the two objectives. This, in conjunction with challenges to the assumption that problems are caused by a few problem animals (Linnell *et al.*, 1999), have led to increased efforts to manage potential conflict proactively using such things as habitat modification, exclusions and repellents. In South Africa cheetah (*Acinonyx jubatus*) are protected under legislation and consequently, instead of trapping and shooting animals that prey on livestock, there is a programme of translocation to fenced reserves, possibly for future ecotourism, although post-release survival rates can be low, especially in the presence of competing predators at release sites (Marnewick *et al.*, 2009). There are also attempts to shift public attitudes to accept lethal control by moving away from a focus on individual animal welfare and towards an appreciation of the need for population level conservation (Linnell *et al.*, 1997).

### *Commercial and recreational*

Commercially and recreationally motivated translocations are often indistinguishable, with introductions to support recreational hunting or derived from the pet trade having a significant commercial element. One of the most significant threats to the conservation of freshwater fishes is the introduction of alien invasive sport fish, with three of the hundred 'World's Worst Invasive Alien Species' being fish introduced solely for sport (Cambray, 2003). One of the earliest examples of fish translocation for stocking a lake was in Turkey by Murat the III, Sultan of Ottoman between 1546 and 1595, and since the 1950s in Turkey a total of 25 exotic fish species have been introduced and 14 native fish species translocated to habitats outside their natural range (Innal & Erk'akan, 2006). There is a globalization of alien fish for sport; for example rainbow trout (*Onchorhynchus mykiss*) are now established in 82 countries (Cambray, 2003). Mammals too are translocated for sport across national and international boundaries. For example, there is evidence of the illegal translocation of invasive feral pigs (*Sus scrofa*) by recreational hunters in Australia to supplement existing populations and to create new populations (Spencer & Hampton, 2005). Red foxes (*Vulpes vulpes*), grey foxes (*Urocyon cinereoargenteus*) and coyotes (*Canis latrans*) are illegally translocated to stock hunting enclosures or 'fox-pens' (Davidson *et al.*, 1992), and in one year private hunting clubs in Kentucky, USA, translocated >2 300 raccoons (Nettles *et al.*, 1979). Ungulate introductions have taken place in well over 50 countries, with the USA and South Africa having the highest numbers of introduced ungulates globally (Spear & Chown, 2009a). To date, however, with only limited evidence of negative impacts, efforts to restrict ungulate introductions may be constrained in the face of the economic gains through trophy hunting and ecotourism (Spear & Chown, 2009b).

The pet trade per se does not fit the definition of translocation because the end point is not intended to be the 'free release' of animals, but it is worth acknowledging that accidental releases and intentional releases by owners can be an endpoint. It has been estimated that in the United States 38 % of exotic bird species are established or establishing from pets that have escaped (Temple, 1992), and two of three invasive bird species in Hong Kong originate from the pet trade (Shieh *et al*., 2006). Since 1994, 290 exotic species of pet birds have been imported into Taiwan, and of these 93 species have escaped and at least 28 of these breed in the wild (Shieh *et al*., 2006).

### *Biological control*

Biological control, or biocontrol, uses living organisms as pest control agents. Classical or traditional biological control involves the introduction of exotic natural enemies to control exotic pests under the simple premise that an exotic organism becomes a pest partly because it has been released from population regulation by natural enemies (Hoddle, 2004). As a form of introduction, therefore, all the risks and uncertainties of potential invasive species apply (Simberloff & Stiling, 1996). Opponents of biological control point to examples where the introduced agent has had an impact on non-target native species, through competition and predation, and other more complex interactions (Simberloff & Stiling, 1996). Proponents of biological control point out that the greatest problem arose in early programmes involving the introduction of generalist vertebrate predators, such as mosquito fish (*Gambusia affinis*), now in about 70 countries worldwide, cane toads (*Bufo marinus*) to target various invertebrate crops pests in Australia, and red foxes and mustelids to control rabbits (*Oryctolagus cuniculus*) in Australia and New Zealand, and that since then increased regulation has reduced risks (Thomas & Willis, 1998; Hoddle, 2004). Increasingly, biological control is being considered for conservation as well as for agriculture (Henneman & Memmott, 2001) and there are examples of proposed development of biological control, such as genetically engineered viral and bacterial diseases, to target invasive arthropods that threaten native flora and fauna (Hoddle, 2004).

### *Aesthetic*

Although less common now, the introduction of exotic species, mostly birds, by Western colonists in the 18th and 19th centuries was a major form of species translocation. Settlers from Europe attempted to create a huntable resource of familiar species, or sought to create new populations of exotic songbirds and other species that were well known to them for purely aesthetic reasons (Duncan *et al*., 2003). Around 70 % of bird introductions (953 events) have been to islands, and over half to Pacific Islands (271 events) and Australasian regions (216 events) (Blackburn & Duncan, 2001). In New Zealand as many as 137 exotic bird species were introduced before 1907, with 284 releases of 43 species occurring mainly between 1861 and 1885 (Duncan, 1997); 28 (20%) of these established populations persist to the present day (Veltman *et al*., 1996). Translocation of animals to support both consumptive and non-consumptive nature-based tourism addresses several motivations, including aesthetic, recreational, education and advocacy, and commercial, as well as conservation (Cousins *et al*., 2008; Mbaiwa, 2008; see Box 1.2).

Box 1.2 **Game ranching in Southern Africa: commercial or conservation translocations?**

Traditionally, the conservation of species, ecosystems and their underlying functions has been accomplished by setting land aside for conservation purposes – the so-called Yellowstone paradigm. In sub-Saharan Africa, as elsewhere, the main shortcoming of the Yellowstone paradigm was the fact that it ignored the vast majority of land that falls outside National Parks (Child, 2000). The generic term 'game ranching', which here includes private game reserves, embodies what has been referred to as a 'second conservation paradigm', which is based on sustainable use, wildlife ownership and pricing (Child, 1996, in Child, 2000). Game ranching, as practised in South Africa (see Figure 1.2), Zimbabwe and Namibia, is defined as the managed, extensive production of free-living animals on large tracts of land that are fenced or unfenced for purposes of live animal sales, hunting, trophy hunting, venison production, tourism or other uses (Bothma, 2002). Officially proclaimed conservation areas in South Africa total 5.8 % of the country's surface area, with private game ranches estimated to contribute an additional 13 % (ABSA 2003, in Carruthers, 2009).

Figure 1.2 **A southern reedbuck from the Drakensberg region of Kwazulu/Natal, South Africa. This species is commonly used in game ranching. (Photo: W. Maartin Strauss).**

The pivotal role that early ranchers in South Africa played in the conservation of ungulates such as the bontebok *Damaliscus dorcas dorcas*, the black wildebeest *Connochaetes gnou* and the Cape mountain zebra *Equus zebra zebra* is well documented. Although the game ranching industry has expanded significantly in recent decades, the conservation role that it plays today is not clear-cut. In an attempt to increase local diversity and thereby economic viability of game ranches, managers now frequently apply artificial selection in breeding indigenous ungulates. The so-called white and black springboks (*Antidorcas marsupialis*) are, for example, popular animals for translocation and at game auctions their monetary value has, on average, increased by 38.4 % and 20.3 % per year over a 10 year period (van der Merwe *et al.*, 2008). Nevertheless, there is an assumption that game ranching contributes more to biodiversity conservation than other forms of agricultural land use (Bond *et al.*, 2004), despite profit being the primary reason for their establishment (Hearne & McKenzie, 2000). There are no official figures available, but du Toit (2007) estimated that up to 70 000 animals are captured and translocated annually in South Africa, resulting in an estimated turnover, including the value of the captured animals, of between R750 million (US$101 million) and R900 million (US$121 million). Prior to the enactment of recent (2004) laws aimed at regulating and controlling the translocation and introduction of large mammals in South Africa, exotic and/or extralimital ungulate species were translocated and introduced into game ranching areas across the country, resulting in South Africa having the second highest number of introduced ungulates globally (Spear & Chown, 2009b).

Game ranching has undoubtedly contributed to the increase in the number of wild animals in South Africa during the last few decades. Conservation in South Africa is, however, at a crossroads, as the game ranching industry pushes for exemption from all nature conservation regulatory control.

### *Religious*

Approximately 30 % of people of all religions in East Asia believe they can accrue merit by freeing captive animals during ceremonies termed 'prayer animal releases'. These ceremonies are organized by temples using local and exotic animals, mostly birds, supplied by pet stores (Severinghaus & Chi, 1999). With the practice prevalent in Taiwan, Malaysia, Thailand, Cambodia, Vietnam, Honk Kong and Korea, and with some temples organizing as many as 24 release ceremonies per year, the scale of translocations is huge. It was estimated, for example, that 128 000 birds were released in only one year in Taichen City, Taipei (Severinghaus & Chi, 1999).

### *Wildlife rehabilitation*

Capture, care and release of wildlife is a significant and growing practice internationally; for example in Britain some 30 000 to 40 000 wild animal casualties end up in wildlife hospitals, the most common species being the European hedgehog (*Erinaceus europaeus*) (Molony *et al.*, 2006). The most frequent causes of injuries to terrestrial species include collisions with vehicles and domestic animal attacks (Hartup, 1996). For seabirds oiling has been a major threat ever since the start of large-scale transportation of petroleum products by sea, with large-scale seabird mortality due to dumping of tanker waste oil from 1917 and oil spills in 1937 off San Francisco (Carter, 2003) and during WWII (Mezat *et al.*, 2002). Following significant oil spills in the late 1960s, rehabilitation efforts for oiled seabirds developed in the United States (Carter, 2003) and South Africa (Nel *et al.*, 2003). Rehabilitation of oiled seabirds involves capture, transport, cleaning and release, and has been characterized by high failure rates, with in some cases only 1–20 % of birds surviving the first year post-release (Mead, 1997). Post-release survival rates vary, however, with the type and degree of oiling, but also with the species. The highest success rates have been achieved with African penguins (*Spheniscus demersus*), with up to 84 % of processed penguins being released (Nel *et al.*, 2003) and up to 65 % of released penguins being resighted within two years (Underhill *et al.*, 1999). Nearly comparable survival rates have been achieved for little penguins (*Eudyptula minor*) (Goldsworthy *et al.*, 2000).

Although conservation of endangered species is one of the reasons cited for the rehabilitation of marine mammals (Moore *et al.*, 2007), the greatest risks involved in the release of rehabilitated animals is that of disease transmission from captivity to wild populations (Quakenbush *et al.*, 2009). There is general agreement by authorities that the health of wild populations should be a greater concern than the welfare of an individual animal, thus euthanasia is often the best option, but one that carries a significant negative image and risks loss of public support for rehabilitation efforts (Moore *et al.*, 2007). Consequently, there may be public pressure to sustain rehabilitation efforts even for species that have healthy populations for which the rehabilitation of a single animal has no conservation value (Moore *et al.*, 2007) but which poses significant risks to wild populations (Quakenbush *et al.*, 2009). In contrast, the numbers of individual birds involved in major oil spills can be significant at a population conservation level; for example oil spills off the South African coast from the *MV Treasure* in 2000 (Parsons & Underhill, 2005) and the *Apollo Sea* in 1994 (Underhill *et al.*, 1999) resulted in the processing of $\sim$ 10 000 and nearly 20 000 oiled African penguins, respectively, from a world total population around that time of <60 000 pairs (Birdlife International, 2008).

### *Animal rights activism and animal liberations*

While conservation biologists are rightly concerned with animal welfare and the reduction of unnecessary suffering, there is a difference between an individual animal welfare perspective that may motivate activities such as animal rehabilitation and the conservation management of wildlife populations. For the most part these differences do not create problems; for example welfare concerns will dictate that any release of captive animals must ensure that each animal has the skills and behaviours necessary for survival in the wild (Waples & Stagoll, 1997). Different perspectives become problematic, however, where animal welfare becomes animal rights. Animal rights activists are committed, *inter alia*, to the total abolition of use of animals in science, commercial agriculture and sport hunting, and consider as fundamentally wrong any system that views non-human animals as resources to be used by humans (Regan, 1983). There exists a challenging incompatibility between a conservation ethic and animal rights, which some see as a 'highly reductionist view' that focuses exclusively on individual sentient animals (Hutchins, 2008). This can lead to illegal liberations of captive animals that effectively expand the range of introduced species and have detrimental impacts on native fauna (Lewis *et al*., 1999). Furthermore, the released animals can suffer. The liberation of captive-bred furbearers such as mink (*Mustela vison*) from fur farms provides one example. Accounts in the media indicate multiple liberations of groups of up to 6000 captive mink from farms in Canada, USA, UK, Ireland, Finland, the Netherlands and Greece over the last decade. Many of the liberations of mink are attributed or claimed to be the actions of the Animal Liberation Front (ALF). Inevitably, released mink start to die in large numbers soon after release, before survivors can be recovered. Nevertheless, liberationists claim that outside their cages mink have a fighting chance of survival. This is despite overwhelming evidence that most freed mink face a slow death in the wild versus a humane end in captivity.

## Conservation translocations

Where conservation of the target species is the primary objective we can consider *conservation translocations* (Hodder & Bullock, 1997) and ask a new question: 'are animals to be released within the historical distribution range of the species?' Releases within the documented natural range may be classified as translocations for *population restoration*, the implication being that the goal is to recover populations of the species back to some past target state. Releases outside the historical range, but with population conservation as the primary objective, are termed *conservation introductions* (IUCN, 1998) (Box 1.1).

## Population restorations

*Reintroduction* is the release of an organism into an area that was once part of its range but from which it has been extirpated (IUCN, 1987) (Box 1.1). In broadly stated terms the objective of a reintroduction is to re-establish a self-sustaining population of a species within its historic range (Griffith *et al*., 1989), and ideally that population will have a high probability of persistence with minimal or no intervention (Seddon, 1999). Despite some early reintroduction success stories, such as Arabian oryx (*Oryx leucoryx*) in Oman (Stanley Price, 1989) and peregrine falcon (*Falco peregrinus*) in North America (Cade & Burnham, 2003), the failure of other, less well-conceived, reintroduction attempts meant that reintroduction project success rates were low (Griffith *et al*., 1989; Wolf *et al*., 1996). The situation was not helped by a lack of post-release monitoring, which meant that the timing and causes of failures was not known (Seddon *et al*., 2007a). In response to the perceived problems the World Conservation Union (IUCN) Reintroduction Specialist Group (RSG) was formed in 1988 under the auspices of the Species Survival Commission (Stanley Price & Soorae, 2003). One of the first actions of the RSG was the formulation of Guidelines for Reintroductions (IUCN, 1998) in order to improve reintroduction practice; for example the guidelines place emphasis on the identification of release sites within the historic range of the species and acknowledge a need to ensure that previous causes of decline have been addressed, both factors having been shown to strongly influence project outcomes (Fischer & Lindenmayer, 2000). In part due to the actions and outputs of the RSG, improved pre-release planning, care over the selection of founders and the composition of founder groups, release site preparation and detailed post-release monitoring have improved project success rates, at least in the short term (Soorae, 2008). Although assessment of reintroduction success is not straightforward it is useful to think of any project needing to progress through two phases (Armstrong & Seddon, 2008): population establishment, which requires survival of founders, and breeding by founders and their offspring; and population persistence, which may be assessed for taxonomically relevant time frames using population modelling tools (Seddon, 1999).

Meta-analyses of factors contributing to reintroduction success indicate the importance of habitat quality at the release site and the number of individuals released (Germano & Bishop, 2009; Griffith *et al.*, 1989; Wolf *et al.*, 1996, 1998). The effects of habitat quality on reintroduction success have been confirmed in recent experimental studies (Moorhouse *et al.*, 2009). The number of animals released, however, is often correlated with several other factors that may be important prerequisites of success. For example, projects that release the most individuals are usually those that are well funded and well resourced, and that are perceived *a priori* to have the greatest chance of success. In contrast, only few founders are released in short-term projects that do not have significant institutional and community support.

*Re-enforcement* (IUCN, 1998), also termed *restocking* (IUCN, 1987) and *supplementation* (IUCN, 1998), or *augmentation* (Maguire & Servheen, 1992), involves the release of individuals into an existing population of conspecifics (Box 1.1), in order to increase population size and reduce the risks of genetic or demographic collapse due to stochastic effects. Translocations for re-enforcement are used to overcome barriers to natural dispersal from other free-ranging populations (e.g. Gusset *et al.*, 2009), to speed up population growth, or to enhance genetic diversity and avoid inbreeding depression (Jamieson *et al.*, 2006). In some cases ongoing re-enforcement may be required to sustain non-viable free-ranging populations until natural productivity is sufficient to support population growth and persistence. For example, kaki or black stilt (*Himantopus novaezelandiae*) are sustained in the wild through the release of captive-reared birds while habitat restoration measures are being trialled (e.g. Keedwell *et al.*, 2002).

Seddon (2010) poses the question of when does a reintroduction become re-enforcement? While seemingly trivial semantics, this question does relate to a more significant one – that of when to stop releases. There is a substantial body of literature that discusses evaluation of reintroduction success (Fischer & Lindenmayer, 2000; Griffith *et al.*, 1989; Seddon, 1999; Wolf *et al.*, 1996, 1998), and there is now widespread use of population modelling to set re-establishment goals, to define optimal reintroduction strategies and to assess population persistence (Armstrong *et al.*, 2002, 2006; Rout *et al.*, 2009; Schaub *et al.*, 2009). Pre-release target setting considers the number, size and composition of founder cohorts and the efficacy of single versus multiple releases. Nevertheless, post-release monitoring will enable refinement of pre-release models (Armstrong & Davidson, 2006; Armstrong *et al.*, 2007;

Wakamiya & Roy, 2009) and may indicate a low probability of population persistence that could be addressed through the release of more individuals. Such *post hoc* secondary releases have been termed *follow-up translocations* (Armstrong & Ewen, 2001) and could be seen as supplementation of the re-established free-ranging population, but should strictly be considered part of the original, but not yet successful, reintroduction attempt (Seddon, 2010).

## Conservation introductions

Mediated movement of organisms outside their native range constitutes a species introduction (IUCN, 1987) and if the goal is the establishment of a new population explicitly and primarily for conservation, then such a translocation is regarded as a *conservation*, or *benign* (in intent at least), *introduction* (IUCN, 1998). The current IUCN guidelines consider conservation introductions to be justified 'when there is no remaining area left within a species' historic range' (IUCN, 1998). This limited rationale marks conservation introductions as a somewhat reactive, stop-gap measure, in some cases perhaps to mark time until appropriate habitat restoration can take place within the historical distribution range of the target species. However, more pro-active interventions are now being considered by natural resource managers, and we can broadly define two types of conservation introduction: *ecological replacement* and *assisted colonization*.

*Ecological replacement* is the release of species outside their historic range in order to fill an ecological niche left vacant by the extinction of a native species. Extinction removes the option of reintroduction through the release of either wild or captive individuals and may mean the loss of critical or otherwise desirable ecological functions. One option is therefore to restore lost ecological function through the establishment of a viable population of an ecologically similar species (Atkinson, 2001). The most readily acceptable approach will be the release of a *subspecific substitute*, such as using the North African subspecies of ostrich (*Struthio camelus camelus*) as a replacement for the extinct Arabian subspecies *S. c. syriacus* (Seddon & Soorae, 1999). The recent use of other *analogue species* includes yellow-crowned night heron (*Nycticorax violacea*) for an extinct endemic *Nyctocorax* species in Bermuda, tundra musk ox (*Ovibos moschatus*) for the extinct *O. palantis* in Sibera (review in Parker *et al*., 2010) and North Island kokako (*Callaeas wilsoni*) for the extinct South Island form *C. cinerea* (see Box 1.3). While subspecific substitutes may be expected to be the most appropriate ecological replacements, other forms may potentially be better functional equivalents. For example, Parker *et al*., (2010) make a case for the replacement of the extinct New Zealand quail (*Coturnix novaezelandiae*), not with its closest extant relative, the Australian stubble quail *C. pectoralis*, but with the more distantly related but ecologically better suited Australian brown quail *C. ypsilophora*. It may not be the case that the analogue species is rare or threatened in its natural range and thus its conservation may not be a primary objective of its introduction as an ecological replacement, necessitating a broader interpretation of the earlier dichotomy between conservation and non-conservation translocations to include the conservation objective beyond the target species (Figure 1.1).

Box 1.3 **North Island kokako translocation to the South Island as an example of an ecological replacement**

In October 2008, 10 North Island kokako *Callaeas wilsoni* were translocated from Mapara in the central North Island of New Zealand to 8 140 ha Secretary Island on the southwestern corner of the South Island, 1 000 km south (see Figure 1.3). South Island forests were previously occupied by a southern kokako species *Callaeas cinerea*, declared extinct in 2004. The intention of the release was to restore the ecological functions of kokako into a South Island forested ecosystem. It is inherently experimental.

IUCN guidelines (1995) and Seddon & Soorae (1999) suggested that an ecological substitute should be selected from extant subspecies or races (rather than species) to avoid fundamental differences in habitat preferences between the original and substitute taxa. However, North and South Island kokako were regarded as subspecies until recently (Holdaway *et al*., 2001). Plumage of the two is the same although South Island birds had orange wattles (small fleshy appendages arising from the gape and lying against the throat) while North Island wattles are blue (Higgins *et al*., 2006). Subtly different behaviours, such as more ground-feeding, may have led to the early decline and extinction of the South Island form due to predation by introduced pest mammals, but this is unclear (Clout & Hay, 1981; Holdaway & Worthy, 1997). The extinction of *C. cinerea* prevents further comparison of behaviours of the two taxa.

Figure 1.3 **A North Island kokako nestling being banded for monitoring purposes on Tiritiri Matangi Island, New Zealand. (Photo: John G. Ewen).**

Diet and behaviour of *C. wilsoni* are very well known (Higgins *et al*., 2006), whereas *C. cinerea* was never studied in detail. *C. wilsoni* mainly eat leaves and fruits, and some insects (Higgins *et al*., 2006). Kokako were very abundant in both islands before human settlement, and their ecological roles included herbivory, pollination and fruit dispersal, as well as being prey for New Zealand's original predators (raptors), some of which are also extinct. The demise of native birds has in turn impaired pollination and perhaps seed dispersal of trees and shrubs (Kelly *et al*., 2010). Kokako are quite large (38 cm; 230 g) with 13 mm gapes, capable of dispersing fruits of several structurally important large-seeded plants (Clout & Hay, 1989; Kelly *et al*., 2010). Reintroducing kokako to Secretary Island is a small part of the biotic restoration planned there. While this translocation meets most objectives of a reintroduction listed by IUCN (1995) – to enhance the long-term survival of a species; to re-establish a keystone species (in the ecological or cultural sense) in an ecosystem; to maintain and/or restore natural biodiversity; to provide long-term economic benefits to the local and/or national economy; to promote conservation awareness – it is primarily an attempt at ecological restoration of a lost biotic community, as first championed in New Zealand by Atkinson (1988).

Monitoring survival of some of the kokako released in October 2008 with transmitters revealed only one death, due to falcon *(Falco novaeseelandiae)* predation. The monitored birds settled in the general area of the island where they were released. A further 17 kokako released in 2009 were sourced from two additional source populations – Kaharoa and Rotoehu – to increase the genetic representation of the North Island species and minimize future inbreeding.

*Assisted colonization*, also referred to as *assisted migration* (McLachlan *et al*., 2007) and *managed relocation* (Richardson *et al*., 2009) has been best defined as 'translocation of a species to favourable habitat beyond their native range to protect them from human induced threats' (Ricciardi & Simberlof, 2009a). Recent interest in this form of conservation introduction has been driven by predicted habitat change due to rapid climate change (Hoegh-Guldberg *et al*., 2008), but assisted colonization could be and has been used to mitigate a variety of threats, including agricultural expansion and urbanization (Ricketts & Imhoff, 2003), filling of hydroelectric reservoirs (Richard-Hansen *et al*., 2000) and the threats posed by (other) deliberately introduced species (Vitousek *et al*., 1997). Specific examples include the translocation of slow-worm (*Anguis fragilis*) from sites for future housing development in the UK (Platenberg & Griffiths, 1999) and giant land snails (*Powelliphanta augusta*) from sites designated for coal mining in North Westland, New Zealand (Walker *et al*., 2008), both of which involved releases outside the species' known historical range.

The debate around assisted colonization has focused on the risk of impacts of introduced species (Mueller & Hellmann, 2008; Ricciardi & Simberlof, 2009a, 2009b; Sax *et al*., 2009; Seddon *et al*., 2009; Vitt *et al*., 2009) and is assumed by many commentators to mark a major shift in conservation translocation practice. However, assisted colonization is a well-established (if previously unnamed) conservation tool in some parts of the world. For example, in New Zealand the extinction threats to endemic birds, herptiles and invertebrates posed by introduced mammalian predators have been addressed with some success through 'marooning', whereby species are translocated to predator-free offshore islands (Saunders & Norton, 2001). In many cases these islands were not within historically documented parts of the species range. In fact one of the earliest examples of assisted colonization could be the pioneering conservation work undertaken by Richard Henry in New Zealand (Jones & Merton, this volume, Chapter 2).

During the 1890s Richard Henry was caretaker of Resolution Island in remote and rugged Fiordland on the west coast of New Zealand's South Island. A keen naturalist, he noted with dismay the impact on native birds of the arrival of recently introduced stoats (*Mustela erminea*) as they invaded this last corner of New Zealand. In a desperate attempt to protect populations of the flightless kakapo (*Strigops habroptilus*) and little spotted kiwi (*Apteryx oweni*) between 1894 and 1900 he translocated hundreds of individuals from the mainland on to Resolution Island (Saunders & Norton, 2001). Unfortunately Resolution was too close to the mainland and stoats invaded in 1900 and Henry's efforts were in vain (Clout, 2006). Nevertheless, the technique of marooning vulnerable species on predator-free islands that may or may not have been occupied by the species in the past became a vital tool to avert extinctions in the face of predation by introduced mammalian predators in New Zealand (Saunders & Norton, 2001).

We can therefore envisage a simple dichotomy between ecological replacement and assisted colonization (Figure 1.1). If the aim is to fill an available ecological niche left vacant by the extinction of the original form, then the type of conservation translocation is an ecological replacement. In the case of marooning and other translocations to move members of a species outside their range to avoid some threat, the primary aim is not to fill an available ecological niche but rather to sustain a viable population, perhaps until appropriate habitat restoration has taken place within their core distribution. In the specific case of climate change mitigation, where a suitable habitat has opened up outside the historic distribution range, then assisted colonization may be used to fill a newly available niche that has not been created by an extinction and is thus similarly distinguished from the release of ecological substitutes for an extinct form (Figure 1.1). In all cases assisted colonization is the human-mediated movement of individuals of a species that would otherwise be unable to survive current or anticipated threats within its current distribution.

## Human dimensions in animal translocations

It is now well understood that intensive conservation interventions, such as reintroductions, cannot hope to succeed without some level of engagement with local and national government, non-governmental agencies and professionals, and, critically, the public (Kleiman, 1989). This is seen nowhere more strongly than in the restoration of populations of large carnivores, where local community attitudes and behaviours can be significant determinants of project outcomes (Hayward *et al*., 2007; Hunter *et al*., 2007; Jule *et al*., 2008; Lohr *et al*., 1996; Meadow *et al*., 2005; Nilsen *et al*., 2007; Zahniser & Singh, 2004).

Public engagement with a translocation project can come at different levels and stages, including the provision of funding (and conversely through economic benefits that accrue to local communities (Kleiman, 1989) and labour, support for approvals, advocacy and lobbying for political and legislative change, and wider attitudinal changes through education and interaction (Williams *et al*., 2002). In many cases community engagement is not just a useful part of translocation planning, nor even just a prerequisite for success; rather it is one of the desirable outcomes.

Translocation projects provide a means to engage with the public, to make them collaborators in the programme (Kleiman, 1989) and to potentially change negative or biased public views of scientists and resource managers. The opportunity to learn about conservation projects, to see wild animals up close and even to participate in their liberation into new areas provides a powerful means to counter the view that conservation is preservation and entails the locking up of resources. Meaningful and positive contact with native species and natural areas can mitigate the alienation from the natural world and the extinction of experience that is a feature of increasingly urbanized societies. It has been suggested that advocacy for the natural world may be the main role of conservation biology (Brussard & Tull, 2007). Parker (2008) proposes that we view translocations as a vehicle to enhance linkages between scientists, managers and the general public, and that meaningful community participation should be considered one of the main outputs of a translocation, along with its management and scientific objectives.

## Concluding comments

Translocation is now well entrenched as a conservation tool, with the numbers of animals being released in reintroduction and re-enforcement projects increasing almost exponentially each year. The more conservation translocation activity there is, the greater the proliferation of terms and concepts being used to describe the various actions, resulting in a mass of synonyms and variations that may end up obscuring meaning. What we have tried to do in this chapter is to provide a standardized and justified terminology to describe the full spectrum of conservation translocation activities. Our hope is that these will enable practitioners and researchers to be very clear about what they are doing or what they propose to do, not only for themselves, but also in discussions or debate with others.

Resource managers have been forced to deal with critical population declines and extinctions, ecosystem degradation and changing habitat conditions due to global climate change (Sekercioglu *et al.*, 2008); consequently new forms of conservation interventions are being explored. These include ecological replacements, whereby functionally equivalent taxa fill a niche left available by the extinction of the original form, and assisted colonization, where species are moved into areas not previously occupied in order to avoid some human-induced threat. These conservation introductions are perhaps not as radical as they first appear, but they do mark the start down an interesting pathway leading away from strict reintroductions towards more controversial decisions about which species we want to have where. There is a clear convergence in thinking between the disciplines of reintroduction biology and restoration ecology (Lipsey & Child, 2007; Seddon *et al.*, 2007b), whereby historical restoration targets are seen as arbitrary and unrealistic, and increasingly there is talk of futuristic restoration (Choi, 2004) and novel (Figure 1.1) or designer ecosystems (Temperton, 2007; Seddon, 2010). The debates will no doubt rage for decades as we consider what our future natural world could or should look like.

## Acknowledgements

This chapter was improved by the comments of Doug Armstrong, Tim Blackburn, John Ewen, Ian Flux, Richard Maloney, Ollie Overdyck, Kevin Parker, Francois Sarrazin, Yolanda van Heezik and Megan Willans.

## References


Armstrong, D.P. & Davidson, R.S. (2006) Developing population models for guiding reintroductions of extirpated bird species back to the New Zealand mainland. *New Zealand Journal of Ecology*, 30, 73–85.

Armstrong, D.P. & Ewen, J.G. (2001) Assessing the value of follow-up translocations: a case study using New Zealand robins. *Biological Conservation*, 101, 239–247.

Armstrong, D.P. & Seddon, P.J. (2008) Directions in reintroduction biology. *Trends in Ecology and Evolution*, 23, 20–25.

Armstrong, D.P., Castro, I. & Griffiths, R. (2007) Using adaptive management to determine requirements of reintroduced populations: the case of the New Zealand hihi. *Journal of Applied Ecology*, 44, 953–962.

Armstrong, D.P., Davidson, R.S., Dimond, W.J. *et al*. (2002) Population dynamics of reintroduced forest birds on New Zealand islands. *Journal of Biogeography*, 29, 609–621.

Armstrong, D.P., Raeburn, E.H., Lewis, R.M. *et al*. (2006) Estimating the viability of a reintroduced New Zealand robin population as a function of predator control. *Journal of Wildlife Management*, 70, 1020–1027.

Atkinson, I.A.E. (1988) Presidential address: opportunities for ecological restoration. *New Zealand Journal of Ecology*, 11, 1–12.

Atkinson, I.A.E. (2001) Introduced mammals and models for restoration. *Biological Conservation*, 99, 81–96.

BirdLife International (2008) Species factsheet: *Spheniscus demersus*. Downloaded from http://www.birdlife.org on 9/9/2009.

Blackburn, T.M. & Duncan, R.P. (2001) Determinants of establishment success in introduced birds. *Nature*, 414, 195–197.

Bond, I., Child, B., Harpe, D. de la *et al*. (2004) Private land contribution to conservation in South Africa. In *Parks in Transition: biodiversity, rural development and the bottom line*, ed. B. Child. IUCN, SASUSG and Earthscan, London, Sterling, Virginia.

Bothma, J. du P. (2002) *Game Ranch Management*. Van Schaik Publishers, Pretoria, South Africa.

Bradley, E.H., Pletshcer, D.H., Bangs, E.E. *et al*. (2005) Evaluating wolf translocation as a nonlethal method to reduce livestock conflicts in the Northwestern United States. *Conservation Biology*, 19, 1498–1508.

Brussard, P.F. & Tull, J.C. (2007) Conservation biology and four types of advocacy. *Conservation Biology*, 21, 21–24.

Cade, T.J. & Burnham, W. (2003). *Return of the Peregrine*. The Peregrine Fund, Boise, Idaho.

Cambray, J.A. (2003) Impact on indigenous species biodiversity caused by the globalisation of alien recreational freshwater fisheries. *Hydrobiologia*, 500, 217–230.

Carruthers, J. (2009) 'Wilding the farm or farming the wild?' The evolution of scientific game ranching in South Africa from the 1960s to the present. *Transactions of the Royal Society of South Africa*, 63(2), 160–181.

Carter, H.R. (2003) Oil and California's seabirds: an overview. *Marine Ornithology*, 31, 1–7.

Child, B. (2000) Making wildlife pay: converting wildlife's comparative advantage into real incentives for having wildlife in African savannas, case studies from Zimbabwe and Zambia. In: *Wildlife Conservation by Sustainable Use*, eds H.H.T. Prins, J.G. Grootenhuis & T.T. Dolan, pp. 335–387. Kluwer Academic Publishers, Massachusetts.

Choi, Y.D. (2004). Theories for ecological restoration in changing environments: Toward 'futuristic' restoration. *Ecological Restoration*, 19, 75–81.

Clout, M.N. (2006) A celebration of kakapo: progress in the conservation of an enigmatic parrot. *Notornis*, 53, 1–2.

Clout, M.N. & Hay, J.R. (1981) South Island kokako (*Callaeas cinerea cinerea*) in Nothofagus forest. *Notornis*, 28, 256–259.

Clout, M.N. & Hay, J.R. (1989) The importance of birds as browsers, pollinators and seed dispersers in New Zealand forests. *New Zealand Journal of Ecology*, 12 (supplement), 27–33.

Cousins, J.A., Sadler, J.P. & Evans, J. (2008) Exploring the role of private wildlife ranching as a conservation tool in South Africa: Stakeholder perspectives. *Ecology and Society*, 13, 43 [online] URL: http://www.ecologyandsociety.org/vol13/iss2/art43/.

Craven, S., Barnes, T. & Kania, G. (1998) Toward a professional position on the translocation of problem wildlife. *Wildlife Society Bulletin*, 26, 171–177.

Davidson, W.R., Appel, M.J., Doster, G.L. *et al.* (1992) Diseases and parasites of red foxes, gray foxes, and coyotes from commercial sources selling to fox-chasing enclosures. *Journal of Wildlife Diseases*, 28, 581–589.

Duncan, R.P. (1997) The role of competition and introduction effort in the success of passeriform birds introduced to New Zealand. *American Naturalist*, 149, 903–915.

Duncan, R.P., Blackburn, T.M. & Sol, D. (2003) The ecology of bird introductions. *Annual Review of Ecology and Evolutionary Systematics*, 34, 71–98.

du Toit, J.G. (2007) Role of the private sector in the wildlife industry. Report, Tshwane, Wildlife Ranching SA/Du Toit Wilddienste. 87 pp.

Fischer, J. & Lindenmayer, D.B. (2000) An assessment of the published results of animal relocations. *Biological Conservation*, 96, 1–11.

Gammons, D.J., Mengak, M.T. & Conner, L.M. (2009) Translocation of nine-banded armadillos. *Human–Wildlife Conflicts*, 3, 64–71.

Germano, J.M. & Bishop, P.J. (2009) Suitability of amphibians and reptiles for translocation. *Conservation Biology*, 23, 7–15.

Goldsworthy, S.D., Giese, M., Gales, R.P. *et al.* (2000) Effects of the *Iron Baron* oil spill on little penguins (*Eudyptula minor*). II. Post-release survival of rehabilitated oiled birds. *Wildlife Research*, 27, 573–582.

Goodrich, J.M. & Miquelle, D.G. (2005) Translocation of problem Amur tigers *Panthera tigris altaica* to alleviate tiger–human conflicts. *Oryx*, 39, 454–457.

Grayson, D.K. (2001) The archaeological record of human impacts on animal populations. *Journal of World Prehistory*, 15, 1–68.

Griffith, B., Scott, J.M., Carpenter, J.W. *et al.* (1989) Translocation as a species conservation tool: status and strategy. *Science*, 245, 477–480.

Gusset, M., Jokoby, O., Muller, M.S. *et al.* (2009) Dogs on the catwalk: modelling re-introduction and translocation of endangered wild dogs in South Africa. *Biological Conservation*, 142, 2774–2781.

Hartup, B.K. (1996) Rehabilitation of native reptiles and amphibians in DuPage County, Illinois. *Journal of Wildlife Diseases*, 32, 109–112.

Hayward, M.W., Adendorff, J., O'Brien, J. *et al.* (2007) Practical considerations for the reintroduction of large, terrestrial, mammalian predators based on reintroductions to South Africa's Eastern Cape Province. *Open Conservation Biology Journal*, 11, 11.

Hearne, J. & McKenzie, M. (2000) Compelling reasons for game ranching in Maputaland. In *Wildlife Conservation by Sustainable Use*, eds H.H.T. Prins, J.G. Grootenhuis & T.T. Dolan, pp. 417–438. Kluwer Academic Publishers, Massachusetts.

Heinsohn, T.E. (2001) Human influences on vertebrate zoogeography: animal translocation and biological invasions across and to the east of Wallace's Line. In *Faunal and Floral Migrations and Evolution in SE Asia–Australia*, eds I. Metcalfe, M.B.J. Smith, M. Morwood & I. Davidson, pp. 153–170. Balkema, Rotterdam.

Henneman, M.L. & Memmott, J. (2001) Infiltration of a Hawaiian community by introduced biological control agents. *Science*, 293, 1314–1316.

Herr, J., Schley, L. & Roper, T.J. (2008) Fate of translocated wild-caught and captive-reared stone martens (*Martes foina*). *European Journal of Wildlife Research*, 54, 511–514.

Higgins P.J., Peter J.M. & Cowling S.J. (2006) *Handbook of Australian, New Zealand and Antarctic Birds*, vol. 7, *Boatbill to Starlings*. Oxford University Press.

Hodder, K.H. & Bullock, J.M. (1997) Translocations of native species in the UK: implications for biodiversity. *Journal of Applied Ecology*, 34, 547–565.

Hoddle, M.S. (2004). Restoring balance using exotic species to control invasive exotic species. *Conservation Biology*, 18, 38–49.

Hoegh-Guldberg, O., Hughes, L., McIntyre, S. *et al.* (2008) Assisted colonization and rapid climate change. *Science*, 321, 345–346.

Holdaway R.N. & Worthy T.H. (1997) A reappraisal of the late quaternary fossil vertebrates of Pyramid Valley Swamp, north Canterbury, New Zealand. *New Zealand Journal of Zoology*, 24, 69–121.

Holdaway R.N., Worthy T.H. & Tennyson, A. J. D. (2001) A working list of breeding bird species of the New Zealand region at first human contact. *New Zealand Journal of Zoology*, 28, 119–187.

Hunter, L.T.B., Pretorius, K., Carlisle, L.C. *et al*. (2007) Restoring lions *Panthera leo* to northern KwaZulu-Natal, South Africa: short-term biological and technical success but equivocal long-term conservation. *Oryx*, 41, 196–204.

Hutchins, M. (2008) Animal rights and conservation. *Conservation Biology*, 22, 815–816.

Innal, D. & Erk'akan, F. (2006) Effects of exotic and translocated fish species in the inland waters of Turkey. *Reviews in Fish Biology and Fisheries*, 16, 39–50.

IUCN (World Conservation Union) (1987) IUCN position statement on the translocation of living organisms: introductions, re-introductions, and re-stocking. IUCN, Gland, Switzerland.

IUCN (World Conservation Union) (1995) Guidelines for re-introductions. Annex 6 to the minutes of the 41st meeting of council. Gland, Switzerland.

IUCN (World Conservation Union) (1998) Guidelines for re-introductions. IUCN/SSC Re-introduction Specialist Group, IUCN, Gland, Switzerland and Cambridge, UK.

IWRC (International Wildlife Rehabilitation Council) (2009) What is wildlife rehabilitation? http://www.iwrc-online.org/. Retrieved 7 September 2009.

Jamieson, I.G., Wallis, G.P & Briskie, J.V. (2006) Inbreeding and endangered species management: is New Zealand out of step with the rest of the world? *Conservation Biology*, 20, 38–47.

Jones, C.G. & Merton, D.V. (2011) A tale of two islands: the rescue and recovery of endemic birds in New Zealand and Mauritius. In Reintroduction Biology: integrating science and management, eds J.G. Ewen, D.P. Armstrong, K.A. Parker & P.J. Seddon, Chapter 2. Wiley-Blackwell, Oxford, UK.

Jule, K.R., Leaver, L.A. & Lea, S.E.G. (2008) The effects of captive experience on reintroduction survival in carnivores: a review and analysis. *Biological Conservation*, 141, 355–363.

Keedwell, R.J., Maloney, R.F. & Murray, D.P. (2002) Predator control for protecting kaki (*Himantopus novaezelandiae*): lessons from 20 years of management. *Biological Conservation*, 105, 369–374.

Kelly D., Ladley J.J., Robertson A.W. *et al*. (2010) Mutualisms with the wreckage of an avifauna: the status of bird pollination and fruit-dispersal in New Zealand. *New Zealand Journal of Ecology*, 34, 66–85.

Kirch, P.V. (2005) Archaeology and global change: the Holocene record. *Annual Review of Environmental Resources*, 30, 409–440.

Kleiman, D.G. (1989) Reintroduction of captive mammals for conservation. *Bioscience*, 39, 152–161

Le Gouar, P., Mihoub, J.-B. & Sarrazin, F. (2011) Dispersal and habitat selection: behavioural and spatial constraints for animal translocations. In Reintroduction Biology: integrating science and management, eds J.G. Ewen, D.P. Armstrong, K.A. Parker & P.J. Seddon. Wiley-Blackwell, Oxford, UK.

Lenain, D.M. & Warrington, S. (2001) Is translocation an effective tool to remove predatory foxes from a desert protected area? *Journal of Arid Environments*, 48, 205–209.

Lewis, J.C., Sallee, K.L. & Golightly Jr, R.T. (1999). Introduction and range expansion of nonnative red foxes (*Vulpes vulpes*) in California. *American Midland Naturalist*, 142, 372–381.

Linnell, J.D.C., Annes, R., Swenson, J.E. *et al*. (1997) Translocation of carnivores as a method for managing problem animals: a review. *Biodiversity and Conservation*, 6, 1245–1257.

Linnell, J.D.C., Odden, J., Smith, M.E. *et al*. (1999) Large carnivores that kill livestock: do 'problem individuals' really exist? *Wildlife Society Bulletin*, 27, 698–705.

Lipsey, M.K. & Child, M.F. (2007) Reintroduction biology and restoration ecology: are two disciplines better than one? *Conservation Biology*, 21, 1387–1388.

Lohr, C., Ballard, W.B. & Bath, A. (1996) Attitudes toward gray wolf reintroduction to New Brunswick. *Wildlife Society Bulletin*, 24, 414–420.

MacLachlan, J.S., Hellmann, J.J. & Schwartz, M.W. (2007) A framework for debate of assisted migration in an era of climate change. *Conservation Biology*, 21, 297–302.

Maguire, L.A. & Servheen, C. (1992) Integrating biological and social concerns in endangered species management: augmentation of grizzly bear populations. *Conservation Biology*, 6, 426–434.

Marnewick, K., Hayward, M.W., Cilliers, D. *et al*. (2009) Survival of cheetahs relocated from ranchland to fenced protected areas in South Africa. In *Reintroduction of Top-Order Predators*, eds M. Hayward & M. Somers. Wiley-Blackwell Publishing, Oxford, UK.

Mbaiwa, J.E. (2008) The success of consumptive wildlife tourism in Africa. In *Tourism and the Consumption of Wildlife*, ed. B. Lovelock, pp. 141–154. Routledge, London and New York.

Mead, C. (1997) Poor prospects for oiled birds. *Nature*, 390, 449–450.

Meadow, R., Reading, R.P., Phillips, M. *et al*. (2005) The influence of persuasive arguments on public attitudes toward a proposed wolf restoration in the southern Rockies. *Wildlife Society Bulletin*, 33, 154–163.

Mezat, J.A.K., Newman, S.H., Gilardi, K.V.K. *et al*. (2002) Advances in oiled bird emergency medicine and management. *Journal of Avian Medicine and Surgery*, 16, 146–149.

Molony, S.E., Dowding, C.V., Baker, P.J. *et al*. (2006) The effect of translocation and temporary captivity on wildlife rehabilitation success: an experimental study using European hedgehogs (*Erinaceus europaeus*). *Biological Conservation*, 130, 530–537.

Moore, M., Early, G., Touhey, K. *et al*. (2007) Rehabilitation and release of marine mammals in the United States: risks and benefits. *Marine Mammal Science*, 23, 731–750.

Moorhouse, T.P., Gelling, M. & Macdonald, D.W. (2009) Effects of habitat quality upon reintroduction success in water voles: evidence from a replicated experiment. *Biological Conservation*, 142, 53–60.

Mueller, J.M. & Hellmann, J.J. (2008) An assessment of invasion risk from assisted migration. *Conservation Biology*, 22, 562–567.

Nel, D.C., Crawford, R.J.M. & Parsons, N. (2003) The conservation status and impact of oiling on the African Penguin. In *Rehabilitation of Oiled African Penguins: a conservation success story*, eds D.C. Nel & P.A. Whittington, pp. 1–7. Birdlife South Africa and the Avian Demography Unit, Cape Town, South Africa.

Nettles, V.F., Shaddock, J.H., Sikes, R.K. *et al*. (1979) Rabies in translocated raccoons. *American Journal of Public Health*, 69, 601–602.

Nilsen, E.B., Milner-Gulland, E.J., Schofield, L. *et al*. (2007) Wolf reintroduction to Scotland: public attitudes and consequences for deer management. *Proceedings of the Royal Society B*, 274, 995–1002.

O'Donnell, M.A. & DeNicola, A.J. (2006) Den site selection of lactating female raccoons following removal and exclusion from suburban residences. *Wildlife Society Bulletin*, 34, 366–370.

Parker, K.A. (2008) Translocations: providing outcomes for wildlife, resource managers, scientists, and the human community. *Restoration Ecology*, 16, 204–209.

Parker, K.A., Seabrook-Davison, M. & Ewen, J.G. (2010) Opportunities for non-native ecological replacements in ecosystem restoration. *Restoration Ecology*, 18, 269–273.

Parsons, N.J. & Underhill, L.G. (2005) Oiled and injured African penguins *Spheniscus demersus* and other seabirds admitted for rehabilitation in the Western Cape, South Africa, 2001 and 2002. *African Journal of Marine Science*, 27, 1–8.

Platenberg, R.J. & Griffiths, R.A. (1999) Translocation of slow-worms (*Anguis fragilis*) as a migration strategy: a case study from south-east England. *Biological Conservation*, 90, 125–132.

Quakenbush, L., Beckmen, K. & Brower, C.D.N. (2009) Rehabilitation and release of marine mammals in the United States: concerns from Alaska. *Marine Mammal Science*, 25, 994–999.

Regan, T. (1983) *The Case for Animal Rights*. University of California Press, Berkeley, California.

Ricciardi, A. & Simberlof, D. (2009a) Assisted colonization is not a viable conservation strategy. *Trends in Ecology and Evolution*, 24, 248–253.

Ricciardi, A. & Simberlof, D. (2009b) Assisted colonization: good intentions and dubious risk assessment. *Trends in Ecology and Evolution*, 24, 476–477.

Richard-Hansen, C., Vie, J.-C. & de Thoisy, B. (2000) Translocation of red howler monkeys (*Alouatta seniculus*) in French Guiana. *Biological Conservation*, 93, 247–253.

Richardson, D.M., Hellmann, J.J., McLachlan, J.S. *et al*. (2009) Multidimensional evaluation of managed relocation. *Proceedings of the National Academy of Science*, 106, 9721–9724.

Ricketts, T. & Imhoff, M. (2003) Biodiversity, urban areas, and agriculture: Locating priority ecoregions for conservation. *Conservation Ecology*, 8(2), 1 [online] URL: http://www.consecol.org/vol8/iss2/art1.

Rout, T.M., Hauser, C.E. & Possingham, H.P. (2009) Optimal adaptive management for the translocation of a threatened species. *Ecological Applications*, 19, 515–526.

Saunders, A. & Norton, D.A. (2001) Ecological restoration at Mainland Islands in New Zealand. *Biological Conservation*, 99, 109–119.

Sax, D.F., Smith, K.F. & Thompson, A.R. (2009) Managed relocation: a nuanced evaluation is needed. *Trends in Ecology and Evolution*, 24, 472–473.

Schaub, M, Zink, R., Beissmann, H. *et al*. (2009) When to end releases in reintroduction programmes: demographic rates and population viability analysis of bearded vultures in the Alps. *Journal of Applied Ecology*, 46, 92–100.

Seddon, P.J. (1999) Persistence without intervention: assessing success in wildlife re-introductions. *Trends in Ecology and Evolution*, 14, 503.

Seddon, P. J. (2010). From reintroduction to assisted colonization: moving along the conservation translocation spectrum. *Restoration Ecology*, 18, 796–802.

Seddon, P.J. & Soorae, P. (1999) Guidelines for subspecific substitutions in wildlife restoration projects. *Conservation Biology*, 13, 177–184.

Seddon, P.J., Armstrong, D.P. & Maloney, R.F. (2007a) Developing the science of reintroduction biology. *Conservation Biology*, 21, 303–312.

Seddon, P.J., Armstrong, D.P. & Maloney, R.F. (2007b) Combining the fields of reintroduction biology and restoration ecology. *Conservation Biology*, 21, 1388–1390.

Seddon, P.J., Armstrong, D.P., Soorae, P. *et al*. (2009) The risks of assisted colonization. *Conservation Biology*, 23, 788–789.

Sekercioglu, C.H., Schneider, S.H., Pay, J.P. *et al*. (2008) Climate change, elevational range shifts and bird extinctions. *Conservation Biology*, 22, 140–150.

SER (Society for Ecological Restoration) International Science and Policy Working Group (2004) *The SER International Primer on Ecological Restoration*. Society for Ecological Restoration International, Tucson. www.ser.org.

Severinghaus, L.L. & Chi, L. (1999) Prayer animal release in Taiwan. *Biological Conservation*, 89, 301–304.

Shieh, B.-S., Lin, Y.-H., Lee, T.-W. *et al.* (2006) Pet trade as sources of introduced bird species in Taiwan. *Taiwania*, 51, 81–86.

Simberloff, D. & Stiling, P. (1996) Risks of species introduced for biological control. *Biological Conservation*, 78, 185–192.

Soorae, P.S. (ed.) (2008) Global re-introduction perspectives: re-introduction case studies from around the globe. IUCN/SSC Re-introduction Specialist Group, Abu Dhabi, UAE.

Spear, D. & Chown, S.L. (2009a) Non-indigenous ungulates as a threat to biodiversity. *Journal of Zoology, London*, 279, 1–17.

Spear, D. & Chown, S.L. (2009b) The extent and impacts of ungulate translocations: South Africa in a global context. *Biological Conservation*, 142, 353–363.

Spencer, P.B.S. & Hampton, J.O. (2005) Illegal translocation and genetic structure of feral pigs in Western Australia. *Journal of Wildlife Management*, 69, 377–384.

Stanley Price, M.R. (1989) *Animal re-introductions: the Arabian oryx in Oman*. Cambridge University Press, Cambridge, UK.

Stanley Price, M.R. & Soorae, P. (2003) Re-introductions: whence and wither? *International Zoo Yearbook*, 38, 61–75.

Temperton, V.M. (2007) The recent double paradigm shift in restoration ecology. *Restoration Ecology*, 15, 344–347.

Temple, S.A. (1992) Exotic birds: a growing problem with no easy solution. *Auk*, 109, 395–397.

Thomas, M.B. & Willis, A.J. (1998) Biocontrol – risky but necessary? *Trends in Ecology and Evolution*, 13, 325–329.

Underhill, L.G., Bartlett, P.A., Baumann, L. *et al.* (1999) Mortality and survival of African Penguins *Spheniscus demersus* involved in the Apollo Sea oil spill: an evaluation of rehabilitation efforts. *Ibis*, 141, 29–37.

van der Merwe, P., Saayman, M. & Krugell, W. (2008) Factors that determine the price of game. *Koedoe*, 47(2), 105–113.

van Vuren, D., Kuenzi, A.J., Loredo, I. *et al.* (1997) Translocation as a non-lethal alternative for managing California ground squirrels. *Journal of Wildlife Management*, 61, 351–359.

Veltman, C.J., Nee, S. & Crawley, M. (1996) Correlates of introduction success in exotic New Zealand birds. *American Naturalist*, 147, 542–557.

Vitousek, P.M., D'Antonio, C.M., Loope, L.L. Rejmanek, M. & Westbrooks, R. (1997) Introduced species: a significant component of human-induced global change. *New Zealand Journal of Ecology*, 21, 1–16.

Vitt, P. Havens, K. & Hoegh-Guldberg, O. (2009) Assisted migration: part of an integrated conservation strategy. *Trends in Ecology and Evolution*, 24, 473–474.

Wakamiya, S.M. & Roy, C.L. (2009) Use of monitoring data and population viability analysis to inform reintroduction decisions: Peregrine falcons in the Midwestern United States. *Biological Conservation*, 142, 1767–1776.

Walker, K.J., Trewick, S.A. & Barker, G.M. (2008) *Powelliphanta augusta*, a new species of land snail, with a description of its former habitat, Stockton coal plateau. *Journal of the Royal Society of New Zealand*, 38, 163–186.

Waples, K.A. & Stagoll, C.S. (1997) Ethical issues in the release of animals from captivity. *Bioscience*, 47, 115–121.

Watson, M. & Thirgood, S. (2001) Could translocation aid hen harrier conservation in the UK? *Animal Conservation*, 4, 37–43.

White, J.P. (2004) Where the wild things are: prehistoric animal translocations in the New Guinea Archipelago. In *Voyages of Discovery: the archaeology of islands*, ed. S.M. FitzPatrick, pp. 147–164. Praeger Publishers, Westport, Connecticut.

Williams, C.K., Ericsson, G. & Heberlein, T.A. (2002) A quantitative summary of attitudes toward wolves and their reintroduction (1972–2000). *Wildlife Society Bulletin*, 30, 575–584.

Wolf, C.M., Garland, T. & Griffith, B. (1998) Predictors of avian and mammalian translocation success: reanalysis with phylogenetically independent contrasts. *Biological Conservation*, 86, 243–255.

Wolf, C.M., Griffith, B., Reed, C. *et al*. (1996) Avian and mammalian translocations: update and reanalysis of 1987 survey data. *Conservation Biology*, 10, 1142–1154.

Zahniser, A. & Singh, A. (2004) Return of the wolves to Yellowstone National Park, USA: a model of ecosystem restoration. *Biodiversity*, 5, 3–7.