Assessing the Viability of Managed Relocation as a Conservation Strategy

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Contents

Acronyms

(AIC)	Akaike's Information Criterion
(AOV)	Analysis of Variance
(CBD)	Convention on Biological Diversity
(CC)	Climate Change
(GDP)	Gross Domestic Product
(GNP)	Gross National Product
(IMF)	International Monetary Fund
(IUCN)	International Union for Conservation of Nature
(MEA)	Millennium Ecosystem Assessment
(MR)	Managed Relocation
(NP)	National Park
(NR)	Nature Reserve
(PR)	Public Relations
(RENEW)	Recovery of Nationally Endangered Wildlife
(RSG)	Reintroduction Specialist Group
(UNDP)	United Nations Development Programme
(UNEP)	United Nations Environment Programme
(WCMC)	World Conservation Monitoring Centre
(WWF)	World Wide Fund for Nature

Abstract

Over recent years it has become increasingly apparent that some species are not able to disperse, or adapt quickly enough, in response to the effects of climate change, and that these taxa face an increased risk of extinction. This has instigated a debate within the conservation community, whether biological units should be intentionally relocated to regions where the likelihood of future persistence is improved. This strategy is known as 'Managed Relocation'. Those on either side of the debate have been appraised, and discussed throughout this study. The financial costs, feasibility, ecological risks and uncertainties associated with managed relocation, must be evaluated, in order to inform future management decisions. This study reviews the cost of 51 mammal relocations, to ascertain the potential economic expenditure of managed relocations. The financial outlay to conduct relocations is broken down into constituent project phases, costs associated with each phase are examined individually. A backward stepwise regression analyses the explanatory power that separate variables have on project expenditure, and descriptive statistics are used to examine various factors influencing the financial costs of relocations. Problems and limitations associated with comparing distinct relocation projects, and using them as a predictive model, are also examined.

Literature that discusses the factors influencing the success and viability of past relocations has been examined, as well as the roles that conventional strategies have to play in preventing climate induced extinctions. Recommendations are made, and identify how current management approaches, and existing environmental and biological modelling techniques, can be adapted to develop an effective managed relocation protocol. An assessment of environmental economics demonstrates that with enhanced efficiency in government spending, and better financial accountability, pecuniary contributions to global conservation initiatives could be substantially increased. This study concludes that information obtained from previous relocations, and environmental modelling, should be used for general guidance, rather than regarded as definitive predictions. In some instances, managed relocation will be deemed unsafe, or too expensive, and conventional management strategies may be more appropriate; but the application of scientific scrutiny will help to reduce risks and financial expense; thus managed relocation has the potential to become a viable conservation strategy.

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1. Introduction

Species movements and mass extinctions have been attributed to major climate events in the past (Gates, 1993; Coope, 1995; Roy *et al*, 1996; Erasmus *et al*, 2002). Geographic ranges of many species are moving to higher altitudes, or towards their respective poles, in response to shifts in habitats to which these species have become adapted; their 'bioclimatic envelope' (Hughes, 2000; Walther *et al*, 2002; Parmesan and Yohe, 2003; Parmesan, 2006; Foden *et al*, 2008). It has become increasingly apparent that some species are not able to disperse, or adapt quickly enough, in response to the effects of climate change (CC) (Mendez *et al*, 2006: Warren, 2006; Hoegh-Guldberg *et al*, 2008). These species face an increased risk of becoming extinct; resulting in many ecosystems ceasing to function in their current form (Pounds *et al*, 1999; Hilbert *et al*, 2001; Hoegh-Guldberg *et al*, 2007; Hoegh-Guldberg *et al*, 2008). Land transformation, habitat destruction and landscape restructuring by humans, has lead to species being unable to naturally disperse throughout their ranges (Thuiller *et al*, 2006). Thomas *et al* (2004) predict that for threatened and endangered species in fragmented habitats, with limited dispersal ability, CC may be the final push towards extinction. This has instigated debate amongst some scientists, resource managers and policy makers, as to whether biological units should be intentionally moved from existing areas of occupancy, to locations which they do not currently inhabit, or have not been known to occur historically, but where it is predicted the likelihood of future persistence will be improved (Hoegh-Guldberg *et al*, 2008; Mueller and Hellmann, 2008; Richardson *et al*. 2009). This practice is referred to as: 'assisted migration', 'assisted colonisation' and 'managed relocation', and from this point will be referred to as managed relocation (MR), the term used by the Managed Relocation Working Group who are assessing its potential as a conservation strategy.

MR is a radical solution developed in response to a pressing problem. Some believe it is a viable method for conserving species and ecosystem functions (Sax *et al*, 2009; Schlaepfer *et al*, 2009; Vitt *et al*, 2009) others disagree, and argue that MR could disrupt ecosystems and cause additional threats (Ricciardi and Simberloff, 2009a and 2009b). The idea of commencing pre-emptive action, to avoid predicted extinction risks, has been bought to the forefront by the International Union for Conservation of Nature (IUCN) who recently assessed species susceptibility to CC impacts (Foden *et al*, 2008). The IUCN report concluded that 70-80% of threatened birds, amphibians and corals are susceptible to CC.

These species are the least resilient to further threats, face an increased risk of extinction, and should be given high conservation priority (Foden *et al*, 2008). Using projected species distributions under future climate scenarios for six sample regions, Thomas *et al* (2004) predict that 15-37% of species would be committed to extinction under mid-range climate warming forecasts for 2050. Levinsky *et al* (2007) assessed 120 terrestrial European mammals and predicted that up to 25% could become critically endangered within 100 years due to CC. Some regions may experience profound climatic shifts within the next 100 years; under these circumstances MR may provide the best chance of survival for some biological units (Hoegh-Guldberg *et al*, 2008). If MR is to become a significant part of integrated conservation strategy, evaluating the benefits and dangers is essential; even if the conclusion proves MR to be an erroneous approach to managing the challenges posed by CC (Sax *et al*, 2009; Schwartz *et al*, 2009; Vitt *et al*, 2009).

Beginning the evaluation process now will allow scientists, resource managers and policy makers to develop informed decisions in the coming years, as CC causes species extinctions to become increasingly common (Sax *et al*, 2009). Schlaepfer *et al* (2009) and Richardson *et al* (2009) recognise the need for a framework that integrates biological and socioeconomic data, however, to develop this framework, a debate on the relative (and subjective) merits of the various courses of action, based on current information, is crucial. Managers are likely to rely on scientists to provide advice as to when MR can be considered an appropriate option. Research specifically focused on MR is required in order for scientists to answer these questions, and provide informed advice (McLachlan *et al*, 2007).

The principle aim of this study is to collect information on the costs of mammal population movements, that have been part of species reintroductions and translocations, to investigate the potential economic expenditure in relation to MR. This will hopefully determine the cost efficiency of MR, and help inform decision makers as to whether MR is an economically feasible option for protecting biodiversity, under certain conditions, as opposed to other conservation strategies. The objective is to collect cost data for individual mammal relocations from project managers, and gather information on different ecological features for each species from 'grey' literature. As well as ecological features, various project location attributes will be assessed, in order to investigate whether any factors have a substantial influence on costs, and should be taken into consideration when evaluating the potential economic feasibility of MR. Previous animal relocation attempts will be reviewed in order to establish how MR could be potentially managed.

2. Background

2.1 Advocates and Opponents of Managed Relocation

If preventing climate driven biodiversity loss is a conservation priority, then MR should be considered as a management option. Even the most optimistic assessments of natural movements present a bleak outlook for most species affected by CC (McLachlan *et al*, 2007). Advocates of MR seek to actively combat the threats posed by CC; proponents of MR believe that climate is the primary influence on the distribution of most taxa, and are confident that future climate projections are accurate, and that species dispersal limitations necessitate human intervention (McLachlan *et al*, 2007). Opponents of MR are wary of the potential for negative impacts as a result of human intervention, although unintentional; and cite the degree of uncertainty as to what drives the distribution of taxa, and the possibility of introducing an invasive pest species, as integral reasons for rejecting MR as a credible conservation management option (Ricciardi and Simberloff, 2009a). Those who believe MR is not a viable solution, emphasise the importance of facilitating natural dispersal, focusing conservation efforts on isolated populations, and preserving landscapes that enable natural population spread (Schwartz *et al*, 2002; Pearson and Dawson, 2005). Those who oppose the use of MR insist that species have accommodated climate shifts in the past (Pitelka *et al*, 1997; Kullman, 1998), but also recognise that relying on natural dispersal may result in the extinction of a species that could otherwise have been saved (McLachlan *et al*, 2007).

The IUCN Reintroduction Specialist Group (RSG) was established in 1988 to address the growing concern that counter-productive relocations were occurring, including the release of species outside their historic ranges (Stanley-Price and Soorae, 2003). The IUCN RSG composed guidelines in order to ensure that relocation planning produced conservation benefits. The guidelines state that ‘benign introduction’ (moving species to suitable habitat outside their normal distribution) is only appropriate when there is no remaining habitat within the historic range (IUCN, 1998). The principle concern of some practitioners is that the expeditious implementation of MR concepts will lead to the inception of ill-conceived relocations (Seddon *et al*, 2009).

Ricciardi and Simberloff (2009a) show particular concern that MR may affect the development and function of ecosystems, by modifying processes such as primary and secondary production, hydrology, nutrient cycling and disturbance. MR may also disrupt ecological interactions, as well as causing an increase in disease and parasites. Sax *et al* (2009) argue that extinctions are permanent, and using MR to reduce the risk of extinction, at a cost of changing ecosystem function and composition, is a viable exchange; especially if most changes are likely to be negligible, rather than detrimental. Vitt *et al* (2009) envisage MR mimicking natural dispersal, which would be expected to occur as a result of shifting climatic envelopes, if ecosystems were still functioning as a geographically continuous landscape; where historically there were few dispersal limitations prior to human disturbance.

Mueller and Hellman (2008) investigated the invasion risk posed by MR, and asserted that species which are subject to intracontinental relocations are less likely to become invasive, as opposed to species that are introduced from outside the recipient continent. According to their findings, although species moved through intracontinental MR are unlikely to become invasive, those species that are potentially invasive are just as likely to cause harm to native species in the recipient region, as species introduced from outside the continent. Nevertheless, existing literature that assesses reintroductions suggests that failing to establish a viable population through MR is more likely to pose a problem, than the risk of unintentionally releasing an invasive species (Wolf *et al*, 1996; Fischer and Lindenmayer, 2000; Van Andel and Grootjans, 2006). Ricciardi and Simberloff (2009a) expound concerns that many MR candidate species lack detailed invasion history, because they have not previously been relocated outside of their historic range. This claim is disputed by MR proponents who assert that many candidate species have had populations restored, or have been bred outside of their native ranges. As a result, ecological restorations have lead to an increased understanding of the aggressive invasiveness of native species (Vitt *et al*, 2009).

Ricciardi and Simberloff (2009b) provide evidence to suggest that mammal, and other species introductions, have contributed to extinctions worldwide (e.g. Clavero and Garcia-Berthou, 2005). Sax *et al* (2009) claim that extinctions caused by invasions of non-native species have mainly occurred on islands, rather than continents, and are primarily a result of predation, rather than competition (Sax and Gaines, 2008). They also insist that scientists and managers are more aware of the risk of causing extinctions through the introduction of invasive species, than opponents of MR often suggest. Schlaepfer *et al* (2009) argue that those who challenge the viability of MR use extreme examples, describing worst-case scenarios, overstating the probability of relocations causing significant damage to recipient ecosystems. Schlaepfer *et al* (2009) highlight that Ricciardi and Simberloff's (2009a) research actually showed that 85% of intracontinental relocations caused no adverse effects on recipient regions, and suggest that extreme examples used by opponents of MR should be balanced by counter examples that reflect the outcome which is most likely to occur.

Ricciardi and Simberloff (2009b) respond to this view with their belief that those who are pro MR are playing down the potential negative consequences; arguing that the examples of plant species introductions, used by Sax *et al* (2009), emphasising that introduced species do not cause extinction of native species, give little information regarding the extent to which the introductions have reduced the populations of native species, altered ecosystem processes and modified habitats. MR opponents contend that introducing non-indigenous species into recipient ecosystems containing close relatives will promote hybridisation that could erode genetic integrity of native populations, and cause the disappearance of distinct species (Ricciardi and Simberloff, 2009a). Vitt *et al* (2009) affirm that MR would ensure that successful evolutionary lineages are given a chance to persist and continue along adaptive evolutionary pathways.

It is perhaps best to approach MR through systematic scientific means; objectively investigating the potential risks and benefits. Through careful planning, adverse effects can be reduced, by restricting harmful actions, monitoring relocations and ensuring management plans are adapted, as knowledge on the subject is developed and improved upon (Hunter, 2007; McLachlan *et al*, 2007; Hoegh-Guldberg *et al*, 2008; Richardson *et al*, 2009). MR proposals call for evidence of imminent threats, quantitative models that predict outcomes, as well as comprehensive strategies. Approaching MR with a constrained view may potentially have a negative impact; as management plans will need to incorporate a substantial amount of data, therefore MR may only be considered for species that have been subject to adequate levels of research. Further problems may be posed if scientific evidence presented to support the implementation of MR is disputed by opponents, causing a delay in conservation action, and potentially leading to further extinctions (McLachlan *et al*, 2007).

Fazey and Fischer (2009) suggest that MR is nothing more than a 'techno-fix' as it does not address the root cause of extinctions, such as human induced CC and habitat fragmentation.

They argue that MR is restricted to treating the symptom of biodiversity loss, and implies that no change in human activities is required. This opinion appears to be somewhat unsubstantiated, as none of the proponents in associated literature imply that MR is a universal remedy for biodiversity loss. Hoegh-Guldberg *et al* (2008) and Vitt *et al* (2009) acknowledge that it is essential for MR to be accompanied by strategies that address the numerous threats that endanger species and their ecosystems, and that MR should be considered a part of integrated conservation strategy. Fazey and Fischer (2009) argue that without concerted efforts to reverse ongoing intensification of production landscapes and fragmentation, managers will become increasingly reliant on MR as the only remaining option. Previous work by Hunter (2007) discusses the danger of exporting problems (i.e. moving species because it is inconvenient to maintain them in their current location when the real threat is not CC), but conservation managers must ensure they are addressing the actual threats affecting species and ecosystems. If CC is identified as a major threat, then in all likelihood it makes sense to act before a species becomes critically endangered.

Fazey and Fischer (2009) state that the widespread adoption of MR could divert resources, efforts and expertise away from ecological restoration projects, and innovative management strategies. McLachlan *et al* (2007) accept that MR can place conservation objectives at odds, but as climate induced threats intensify, practitioners will be required to consider a multitude of management options, and implement the most appropriate solution. Risks exist on both sides of the scale; rejecting MR based on uncertainties could lead to a potentially innovative strategy being prematurely discarded. In order to assess the viability of MR as a conservation policy, centralised implementation is required; as well as careful evaluation of the risks, and a broad debate on the legal, ethical and biological issues.

2.2 Justifying Managed Relocation and Identifying Potential Candidate Species

Until recent years, the process of deciding whether MR should be undertaken has received little attention (Richardson *et al*, 2009). Hoegh-Guldberg *et al* (2008) proposed a stepwise linear decision framework, to assess species suitability for MR, to maximise the benefits of MR, whilst minimising the risk of undesirable outcomes. This framework outlines possible actions required, in response to potential future climate scenarios, and surmises that providing a risk assessment and management framework will enable managers to identify low-risk conditions suitable for MR. In the first instance, Hoegh-Guldberg *et al* (2008) determine the extinction risk that CC poses to a species, if the threat is perceived to be low, the framework suggests continuing conventional conservation strategies is possibly the best approach, if the perceived risk is high, then the feasibility of relocation is assessed. If relocation is not considered appropriate, invoking ex-situ conservation is considered as one available option, the second is to accommodate natural movement through habitat re-creation, accompanied by species protection once a species arrives in the restored ecosystem. However, this is only possible if species are able to naturally disperse, and may result in the necessity to create new habitats for species in areas outside historical ranges, potentially requiring the MR of obligatory resources, essential to accommodate the focal species. The second option also fails to recognise that protecting organisms as they arrive may not be practical; and that habitat recreation and migrant species may not be accommodated by stakeholders and communities in the recipient region.

If MR is deemed appropriate, Hoegh-Guldberg *et al* (2008) suggest that the benefits must outweigh the biological risks and socioeconomic costs, otherwise conventional conservation approaches should be adopted. The drawback of using the Hoegh-Guldberg *et al* (2008) framework, is that species being assessed may not be considered a suitable MR candidate, until all conventional strategies have proved to be ineffectual; resulting in time, resources and expertise being misspent. Furthermore, species may not be considered appropriate candidates, until the risk of extinction becomes great enough that the benefits of MR are justified, by which point, attempts to save the population could become more complex.

Hoegh-Guldberg *et al* (2008) suggest that species confined to highly fragmented ecosystems, or those populating habitats that are declining due to CC, are suitable candidates for MR. Widespread generalist species, and taxa that are able to maintain large populations through natural dispersal, are unlikely to be considered for MR. However, generalist species may be appropriate for MR; if their population contains individuals whose genotypes are preadapted for survival beyond current range limits. These may present little risk to recipient ecosystems if they are similar to the species already occupying the region, although there are some potential problems with this strategy. Selecting preadapted genotypes from the periphery of a population may result in genotypes at the core, potentially most at risk from CC, being overlooked, and becoming extinct. This could lead to a loss of genetic diversity within a population, and therefore undermines the argument that MR preserves evolutionary lineages and adaptive pathways (Hampe and Petit, 2005; Vitt *et al*, 2009). Furthermore, moving individuals that are better adapted for survival into a novel ecosystem, may give them a competitive advantage over the native species. Once immersed in a new environment, preadapted genotypes could experience rapid adaptations in morphology, physiology, behaviour and life history, all of which may affect their predicted impact on the recipient ecosystem (Ricciardi and Simberloff, 2009a).

The Hoegh-Guldberg *et al* (2008) management framework is a creditable attempt at devising an MR protocol, yet the summation appears to be that MR should be considered a last resort, if conventional strategies are found to be ineffective. Richardson *et al* (2009) highlight several drawbacks associated with the Hoegh-Guldberg *et al* (2008) approach. In their view, the multi-dimension decision making process, required to develop conservation management strategies, is not suitable for resolution via a decision tree, because it fails to evaluate the relative merits of competing management options. The Hoegh-Guldberg *et al* (2008) linear decision framework does not consider social and cultural values when assessing the viability of MR, and fails to evaluate competing needs and interests (Richardson *et al*, 2009).

Richardson *et al* (2009) devised a multidimensional framework to evaluate the validity of MR for individual biological units, which prioritises different courses of action, and evaluates the likelihood and consequence of potential impacts, based upon social and ecological criteria. The framework assesses four categories including: 'focal impact' which measures the potential effects that CC and MR have on the focal biological unit and its community, 'collateral impact' considers the effects MR may have on the recipient ecosystem, 'feasibility' gauges the logistical, legal and technical practicalities that may prevent MR from achieving its objective, and 'acceptability' measures peoples tolerance of MR; primarily based on sociology, ethics and social norms. The 'focal' and 'collateral' categories attempt to determine the probability of undesirable effects occurring, these potential risks may be theoretically foreseeable or predictable, but limitations in time and resources are likely to restrict the accuracy of estimated risk assessments (Richardson *et al*, 2009).

The disclosure of information through multidimensional frameworks provides a useful cost-benefit analysis for practitioners, invokes public participation, highlights the potential for alternative strategies, and stimulates the debate relating to MR, which reinforces the legitimacy and acceptability of management decisions, should MR be undertaken (Svancara *et al*, 2005; Regan *et al*, 2006; Camancho, 2007; Richardson *et al*, 2009). However, ecological criteria, expert judgement and available data, will change over time, as new analyses and experiments are conducted, potentially modifying the way frameworks are utilised. Different stakeholder groups are likely to form varying conclusions regarding MR, even when presented with the same information. As more data becomes available, public perception, cultural values, and social norms will undoubtedly shift, and evaluations of MR will vary accordingly (Richardson *et al*, 2009). Therefore, opponents of MR may question the validity of a framework as a robust decision making protocol, and express concern that MR management is based upon social values and information, which are subject to change in the future. McLachlan *et al* (2007) advocate that the growing threat posed by CC requires immediate action, regardless of forthcoming scientific developments; that there is no time to wait for better data, and the issue of climate induced extinctions needs to be addressed based on the existing information.

Sandler (2008) assessed the social perceptions and ethics relating to MR, and investigated how the instrumental, ecological and intrinsic value of a species could be used to justify the potential for MR. Advocates of MR do not promote the instrumental value of species (i.e. their usefulness to humans) as a motivation for implementing MR. Many species are low in such values, as only a small proportion are economically significant, or medicinally functional (Maclaurin and Sterelny, 2008). MR candidate species are likely to be narrowly distributed, or rare, so their contribution to ecosystem services and other instrumental values are likely to be low (Sandler, 2008). Endangered species are often put at risk, because the economic value of extraction activities and development is higher than their instrumental value (Maclaurin and Sterelny, 2008). This suggests that the instrumental value of species is unlikely to act as a sound basis in advocating MR.

A species ecological value is measured by the contribution it makes to ecosystem integrity. MR candidates should present low ecological risks to the recipient regions, therefore, MR candidates are unlikely to offer substantial ecological benefits (Davidson and Simkanin, 2008; and Mueller and Hellman, 2008). The greater the ecological value of a candidate species, the greater the associated risks; taxa that are selected based on their ecological impact, could have a substantial influence upon the functions of a recipient ecosystem. This suggests that ecological value is perhaps not appropriate for justifying MR (Sandler, 2008). Despite this, many ecosystems are no longer considered pristine as some species may have been extirpated. Hunter (2007) discusses the possibility of 'climatic refugees' being used to fulfil the roles of extirpated dominant and keystone species, in order to restore ecosystems to their original state. This scenario proposes that species can be selected for MR, based upon their potential ecological value. However, if a species ecological value alters the state of the recipient ecosystem, MR is likely to be vehemently opposed.

The intrinsic object value of a species, as described by Rolston (1985; 2001), is the unique, and potentially productive evolutionary trajectory, which each individual species represents. On this basis, Rolston advocates that species need to be protected, because extinction will shut down the generative evolutionary process, but intrinsic object value does present problems in the assessment of candidate species. Intrinsic object value is not based on objective facts that are independent of human construction, information used to form management plans needs to be gathered through impartial empirical science, rather than subjective attitudes. Therefore, MR proposals based on intrinsic object value, would not be constructed using information subject to pragmatic scientific scrutiny (Sandler, 2008). Even if intrinsic object value can be assigned to a species; not all organisms are equally distinctive forms, representing potentially productive evolutionary trajectories, therefore the justification for MR would not be equal for all species (Sandler, 2008). Rolston (2001) asserts that species possess intrinsic object value, by virtue of the role they play within evolutionary processes and ecological systems; it is not the preservation of the species itself, but the preservation of the species role within a system, which is important. Therefore it is debatable whether preserving species through MR preserves its intrinsic object value. Although MR may prevent the evolutionary potential and trajectory from being extinguished, it may be immeasurably altered (Sandler, 2008). For these reasons, the intrinsic object value of a species offers insufficient justification for MR to be adopted.

The valuer-dependent approach perhaps offers a more robust basis for justifying MR. If a species is deemed significant, based on its intrinsic-value, it’s worth is justified by, and integrated with, an individual’s worldview (Sandler, 2008). Callicott (1989) suggests that species may not be economically, or ecologically valuable, but their existence is worth preserving; therefore MR may be justified. Sagoff (1988) and Callicott (2006) highlight the fact that the preservation of intrinsic value is often expressed in a legislative context, which is founded through democratic processes, making intrinsic value a strong basis for justifying MR. However, if a species value is established through democratic process, then this value can also be revoked if there are sufficient facts to substantiate revising the basis on which a species is valued (Sandler, 2008). Agriculture, property rights and industry advocates may be able to provide reasons, incorporated in their belief system, to support the revision of a species intrinsic value. In addition, many species exist in countries which lack legislation that recognises the intrinsic value of a species (Sandler, 2008). Furthermore, legislation can place restrictions on management options and land use, which could fuel social resistance to introducing species (Shirey and Lamberti, 2010). On these grounds it could be argued that a valuer-dependent intrinsic approach does not present an entirely robust justification for MR.

Broad agreement on the social and ethical acceptability of MR is unlikely to be achieved, without an objective scientific assessment of the feasibility of potential options to combat climate driven extinctions. Hunter (2007) and Seddon *et al* (2007) propose that addressing the issue of MR indirectly, such as focusing research questions across various disciplines, and filling information voids, a foundation of knowledge for MR will be established. Evaluating technology, humaneness and economic cost will aid the development of flexible management strategies, and potentially increase the validity of MR. Deciding whether or not to undertake MR is evidently problematic. Many issues are not easily gauged, others will undoubtedly arise as knowledge on this subject advances, and it is likely that on some occasions MR will be rejected because it is too hazardous or expensive. As Hunter (2007) and McLachlan *et al* (2007) maintain, science may reduce the risks and costs, and help improve conservation practices, eventually providing a resolution to the uncertainty and debates surrounding MR. This research seeks to aid this debate by using information from previous mammal relocations in order to ascertain the economic feasibility, and the validity of MR as a management option.

3. Methodology

In this study the term 'relocation', as adopted by Fischer and Lindenmayer (2000), is used to define the intentional movement by humans of an animal, or populations of animals, from one location to another. This neutral overarching term is used in order to avoid the confusion that can be caused by other terminologies. If a specific project is referred to within the text the correct terminology for the type of relocation undertaken is used. The IUCN (1998) indentify and define four types of relocation: '*Reintroduction*' is an effort to establish a species in an area which was once part of its historical range, but from which it has been extirpated or is extinct. '*Translocation'* is a deliberate and mediated movement of wild individuals or populations from one part of their range to another. '*Reinforcement /Augmentation / Supplementation'* is the addition of individuals to an existing population of conspecifics. '*Conservation / Benign Introduction'* is an attempt to establish a species, for the purpose of conservation, outside its recorded distribution, within appropriate habitat and ecogeographical area.

This study assesses the economic costs, successes and management of previous relocation projects, and investigates factors that have influenced the economic viability and effectiveness of global mammal relocations. Data detailing project costs were obtained by electronically mailing surveys to relocation project managers. Investigations by Griffith *et al* (1989), Wolfe *et al* (1996) and Fischer and Lindenmayer (2000) adopted this method; mailing surveys to wildlife managers. Their results were used to identify ecological factors that influence the success of previous bird and mammal relocations. Reading *et al* (1997) employed a similar approach when evaluating anthropogenic factors that affect the positive outcomes of a relocation, as did Martins *et al* (2006) when they investigated the associated economic costs of eradicating alien mammals from islands. Existing literature on previous relocations was reviewed in order to investigate issues that have historically affected relocation projects; as these factors could also potentially impact MR effectiveness in preventing climate driven species extinctions.

The cost data and literature review will aid the evaluation of MR as a viable and cost effective management option, as well as assisting in the development of recommendations, outlining how MR could be approached. It was decided from the outset that this study would focus solely on mammal relocations, due to the availability of mammal data (Griffith *et al*, 1989; Wolf *et al*, 1996; Fischer and Lindenmayer, 2000) as well as the greater ecological and morphological variation between species (Hayes and Jenkins, 1997). A list of mammal species that have been subject to relocations, compiled by the IUCN RSG, was supplied by Professor John E. Fa (2010). A mailing list of specialists involved in the relocation of species, identified by the RSG, was assembled by obtaining contact details from academic journals, project reports, internet searches, personal web pages and correspondence with organisations involved with project management. The majority of relocation projects have taken place in North America and the Caribbean, Meso and South America, Western Europe, Oceania and Africa; while fewer have been conducted in Eastern Europe and Asia.

Using Microsoft Excel a data sheet was composed and the various economic costs likely to be incurred by relocation projects were categorised. The data sheet separated the relocation process into eight constituent steps that have been identified in animal relocation literature (e.g. Kleiman *et al,* 1991; Fa *et al,* in press). The eight project steps were listed in separate columns which included: acquiring animals for release, feasibility studies, selecting release site, site preparation , transport of animals, release strategy, post release activities (e.g. monitoring, supplementary feeding) and reporting project outcomes. The data sheet listed potential factors where financial costs may be incurred; these were published in rows next to the columns identifying project components. Factors included: staff salaries, maintaining ex-situ facilities, animal husbandry, capturing individuals from wild populations, veterinary care, transport (e.g. vehicle hire, fuel), equipment, habitat modification and / or restoration, reducing human and wildlife conflicts, publishing reports and literature, and meetings and consultations.

Participants were asked to input the financial outlay for each step in the relevant column and row of the data sheet; identifying where the project costs were incurred, and why they were required. In the few cases where participants supplied combined cost for a single factor, and failed to identify specifically where costs were incurred, the total outlay for that factor was then associated to the appropriate steps, according to where they were most likely to be incurred. These assumptions were based on more detailed datasets provided for other relocations. Participants were invited to add rows if there were additional factors with a financial expense that were not included on the data sheet. It was understood that not all relocation projects would necessarily undertake each of the identified processes, and this was clearly explained. Participants were asked to provide costs in US Dollars ($US) using the exchange rates existing at the time the project was carried out. If this was not possible, participants indicated which currency they used when completing the form, and the currency was converted into $US using the average annual exchange rate over the duration of the project. The average rate was calculated using the exchange rate on the first day of each financial quarter specified on (http://www.x-rates.com /), whose rates are gathered from the Federal Reserve Bank of New York and the International Monetary Fund. Participants were also asked to specify: species involved, country where the relocation took place, year of project commencement, length of time needed to undertake the relocation, number of individuals relocated, and whether species were held in captivity prior to release and for how long. Captivity costs included in the analysis are those directly associated with the relocation project, and exclude any costs relating to ongoing ex-situ conservation programmes, prior to project commencement.

Data sheets were emailed to over six hundred managers, and included a covering letter explaining the purpose of the survey. Data sheets were also circulated via internal mail servers amongst members of the IUCN RSG, Wildlife Translocation Association of South Africa, US Northern Wild Sheep and Goat Council, Federal Government Fish and Wildlife Services across the United States of America, and a number of Zoological Parks. Once the completed data sheets had been returned, it became apparent that some were estimated costs, calculated according to the managers' knowledge of the project. A quality scoring system was used to evaluate each of the data sheets; calculated using the percentage of costs that the managers indicated were an estimate. For example, data sheets with costs based entirely on estimates received a data quality score of zero, while data sheets based on the actual project costs, containing no estimates, were assigned a score of one hundred. If sixty nine percent, or lower, of the costs were estimates, it was treated as an 'estimate' overall. If seventy percent of the data and upwards were based on accurate costs, this was treated as 'actual' cost data. Responses were categorised in this way, in order to investigate whether cost data predominantly based on estimates significantly varied from data based on actual project budgets.

Factors that were identified as potential effects on project costs included: adult body mass, region where the project was undertaken, gross domestic product (GDP), and conservation status; which have been indentified in previous studies (e.g. Peters, 1983; Balmford *et al*, 2003; Fa *et al*, in press) as factors that can impact on conservation management. Data for adult body mass was gathered from the PanTHERIA 1.0 database of life history, ecology and geography of extant and recently extinct mammals (Jones *et al*, 2009). Eight separate regions were identified on the RSG list, including: North America and the Caribbean, Meso and South America, Africa, Western Europe, Eastern Europe and North and Central Asia, West Asia, South and East Asia and Oceania. Data identifying GDP per capita, for the country where the relocation project was conducted, was assembled using the World Economic Outlook Database (International Monetary Fund, 2009). If the project spanned a number of years, the mean annual GDP was calculated over the project duration. Species conservation status was defined using the IUCN category assessment published in the 'Red List of Threatened Species' (IUCN, 2010).

Due the diverse nature of relocation projects, the cost data supplied varied significantly between projects. In order to evaluate potential differences between the relocation programmes, it was necessary to decide upon the most appropriate method of assessing the data in order to identify factors that impacted project costs. It was deemed inappropriate to compare the cost per kilogram of individual animals, this biased cost effectiveness towards species with greater body mass. Comparing the cost per year was not feasible, as project timescales ranged from one week to eighteen years. Forecasting annual costs for shorter projects, based on the data provided, risked increasing the appearance of the total cost of a relocation; impacting on the accuracy of the results. Comparing the total cost for individual relocation projects posed further difficulties, as projects running for a longer period of time often incurred larger costs, such as staff salaries and consumables. Therefore it was decided that cost per individual would provide the best indication of the total pecuniary expenditure required to conduct relocations for a specific species. This methodology also contains limitations and biases, which have been identified through the course of this study, and will be discussed in further detail in the results and discussion chapters. The data had a non-normal distribution so the medians ($\tilde{x}$) of the multiple cost values was used in the data analysis to ensure that outliers at the extreme of the dataset had less of an impact. This was done to ensure that the variance between the cost per individual for the different project phases, and explanatory variables, could be assessed with increased accuracy. In order to effectively analyse the numerical data, values were compressed into a base 10 logarithm. A stepwise regression was carried out using R 2.11.1 statistical software package, and a backward elimination of candidate variables was carried out to identify factors that had a significant effect on the cost of conducting relocations.

4. <u>Results</u>

4.1 Dataset Results

The cost data of fifty one relocations, for a total of thirty eight species, was supplied by thirty three managers; a summary of the dataset is attached in Appendix 1. The dataset is mainly comprised of relocations for species from three orders, these include: sixteen (31%) Artiodactyla, eleven (21%) Carnivora and nine (18%) Rodentia. Of the total relocations thirty five (69%) are reintroductions and sixteen (31%) are translocations. Data sheets for Forty one (80%) relocations were completed by participants, the remainder were compiled using the project budgets provided by managers. The data for thirty seven (73%) relocations is derived from the actual project costs quoted, the remaining twelve (23%) relocations are based on estimated costs provided. The African buffalo, and one of the African elephant translocations, are based on hypothetical financial outlays; using expert knowledge of current translocation costs. Of the forty nine actual relocations, all have been successful in establishing persistent populations, with the exception of the Greater stick-nest rat reintroduction in 2000.

Relocations in scope were carried out in nineteen countries and six regions, Africa accounted for eighteen (35%) of the relocations, eleven (21%) took place in North America/Caribbean, nine (18%) in Oceania, eight (16%) in Western Europe, four (8%) in Meso and South America and one (2%) in South and East Asia. GDP values for Africa, Asia and Meso, and South America range from $400 to $7,000, whereas GDP values in Western Europe, Oceania and North America/Caribbean are higher, and range from $15,000 to $49,000. The lowest GDP was $407 in Vietnam during the Pygmy Loris reintroduction. The highest GDP was $48,950 in Australia with the reintroduction of the Woylie and Bridled nailtail wallaby.

Individual body mass ranged from 0.0014 Kg for the Perdido key beach mouse to 3,824 Kg for the African elephant. The number of individuals released ranged from one, including the American manatee reintroduction and Cheetah translocation, and five hundred of the Burchell's zebra. Project durations varied from eighteen years for the Chacoan peccary, to one week for Mountain goat, Woylie and Bridled nailtail wallaby reintroductions. Most relocations are relatively recent, thirty eight (74%) took place between 2000 and the present, 9 (18%) between 1990 and 1999, and three (6%) between 1980 and 1989. As highlighted by **Table 4.1.1,** the number of days taken for each relocation varied between orders

Table 4.1.1: Difference in number of days needed to undertake relocations for various orders. Sirenia, Dasyuromorphia, Peramelemorphia, Perissodactyla and Proboscidae have not been included, as insufficient data limited the capacity to conduct basic analysis.

Order	n	Mean($\bar{x}$)	Median($\tilde{x}$)	Range
ARTIODACTYLA	16	1008	365	7 – 6570
RODENTIA	9	874	365	60 – 3650
CARNIVORA	11	1715	1642	730 – 2920
PRIMATES	4	706	592	182 – 1460
DIPRODONTIA	4	642	1642	7 – 1642

Table 4.1.2 illustrates the number of relocations conducted within each conservation category. A large proportion of relocations involved species in the 'Least Concern' category, because not all relocations had a conservation objective. Relocations were conducted for tourism purposes, movements of game species, and augmenting populations of species considered regionally, rather than internationally threatened (Childers, 2010; Goldstein, 2010; Myatt, 2010; Wilson, 2010).

Table 4.1.2: Relocations incorporated in the dataset classified using the focal species IUCN Red List classification. (n) The total number of relocations undertaken in the classification, (%) percentage of the dataset the relocations represent, (Number of Species in Category) the number of individual species that are represented in each classification.

IUCN Red List Conservation Status	**n**	**%**	**Number of Species in Category**
Least Concern	19	37	15
Near Threatened	3	6	2
Vulnerable	11	22	9
Endangered	12	23	7
Critically Endangered	5	10	4
Extinct in the Wild	1	2	1

Total project costs ranged from $2,065 for the Lowland sykes monkey, to $2,377,194 for the European otter ($\bar{x}$ = $317,148, $\tilde{x}$ = $155,575). Total cost per individual ranged from $66 for Lowland sykes monkey, to $151,299 for the Brown Bear ($\bar{x}$ = $23,258, $\tilde{x}$ = $6,632). Costs varied between different project phases, with some relocations investing more money on certain activities than others, **Figure 4.1.1** shows the median expenditure for each, and illustrates how the financial outlay was distributed between various project phases.

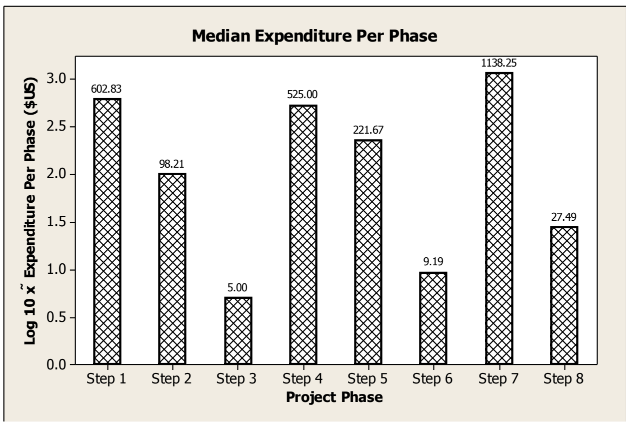

Figure 4.1.1: Log 10 median ($\tilde{x}$) expenditure for each project phase. (Step 1) Acquiring animals for release, (Step 2) Feasibility and background study, (Step 3) Selection of release site, (Step 4) Preparation of release site, (Step 5) Transporting individuals to release site, (Step 6) Release strategy, (Step 7) Post release activities, (Step 8) Reporting and assessment.

4.2 Step 1: 'Acquiring Animals for Release'

A majority of the expense incurred by projects during this phase related to staff wages, maintaining facilities, husbandry, veterinary care, capturing wild individuals, transport and equipment. The 1997 reintroduction of nine African wild dogs incurred the greatest expense in acquiring animals for release, as captive bred dogs were purchased at a cost of $11,412 per individual. These African wild dogs were reintroduced as part of a tourist attraction for a Safari Lodge. Captive bred African wild dogs were also purchased for another three reintroductions, which took place in Nature Reserves (NR) and National Parks (NP), yet the cost per individual was much lower ($1,301, $2,523 and $3,028), it is unclear whether the cost of captive bred animals had fallen, or preferential rates were provided for projects conducted for conservation purposes. Reintroductions reliant on wild caught dogs incurred capture costs between $61 and $334 per individual, and obtained six and eleven dogs respectively. This suggests that capturing wild individuals, in some instances, may be a more cost effective option of acquiring animals for relocations; if a sustainable source population is available. Other projects that purchased captive animals for release were the Eland translocation, and Water vole reintroduction.

Purchasing captive bred animals could potentially increase the cost of conducting MR. Tudge (1992) suggests that commercial breeders could play a role in helping conservation professionals prevent species loss. Yet the data supplied implies that purchasing animals from breeders substantially increases costs, potentially affecting the economic feasibility of MR; thus releasing individuals sourced from wild populations could be perceived as the better option.

The second highest cost during this phase ($11,048 per individual) was encountered by the project reintroducing six Beavers within the UK. The majority of this cost ($10,414) was due to animals being maintained in quarantine for six months prior to release. The Beaver reintroduction in Hungary required no prior captive maintenance, and the cost of acquiring animals was $9 per individual. This suggests that relocations requiring animals to be kept in captivity prior to release, potentially experience increased costs during this phase. If MR is to take place, it would be advisable to assume that prior to relocating species outside their historic range, it may be necessary to quarantine animals, to reduce the risk of releasing parasites or disease into the recipient region. Greater costs could be incurred during this phase, if it is necessary for animals to be maintained in quarantine prior to MR.

Data supplied for Addax and Mountain bongo relocations incurred no costs acquiring animals for release. However, both operations released individuals that had been maintained within zoological parks, substantial costs associated with maintaining species ex-situ preceding reintroduction have been omitted from the data, as participants were asked to only include the costs incurred for keeping animals in captivity when directly associated with the relocation. These figures demonstrate that costs associated with maintaining species in long-term captivity prior to relocation may dramatically impact the economic viability of MR.

The Greater stick-nest rat reintroduction in 2000, and the Red squirrel reintroduction also reported no costs were incurred during this phase, and it remained unclear as to how animals were obtained. However, the cost for acquiring Greater stick-nest rats for the 1998 reintroduction was $1,080 per animal. There were also noticeable differences in costs between the two Mexican howler monkey translocations. Obtaining individuals for the 1988 operation cost $1,852 per animal, compared to $111 in 2002. Rodriguez (2010) explains that this difference was due to the initial financial outlay; materials and facilities being in place for the second translocation, in addition personnel gained more experience during the first translocation, reducing the time required to complete this phase during the latter translocation. These projects emphasise how costs can vary significantly in the relocation of the same species, even in the same region.

It is apparent that using financial records for previous relocations to predict costs for similar future projects invariably has limitations. However, it is encouraging to note that cost effectiveness and efficiency is likely to improve with each relocation managers undertake. The initial outlay for facilities and equipment may be substantial, but these are a one-off cost; for example the same equipment could be used in future relocations, thus reducing costs of subsequent projects. Staff efficiency may also improve with experience, reducing time and salary costs, and potentially increasing the likelihood of a project successfully establishing a relocated population (Rodriguez, 2010).

4.3 Step 2: Feasibility and Background Study

The greatest financial outlay for this phase was encountered by the Brown bear reintroduction, project managers used budgets to provide an overall cost for each factor, but failed to specify individual phases where costs were incurred. Expenses were divided evenly between the factors and phases where managers identified financial outlay had occurred, resulting in an estimated; $10,000 for salaries, $41,000 reducing human/wildlife conflict, $46,166 producing reports and literature, and $46,166 for meetings and consultations, and the estimated cost per individual released for this phase is $14,333.

It could be argued this method provides inaccurate figures, but when this outlay is compared to the reintroduction of nine African wild dogs in 2000, where costs were derived from actual budgets, this estimate gives an indication for costs associated with feasibility studies for large carnivore relocations. The African wild dog relocation was the third most expensive project spending $43,117 on reducing human/wildlife conflict and $18,365 on meetings and consultations, totalling $6,831 per individual. The fact that Brown bears were reintroduced to the eastern Alps, and African wild dogs confined to a NR, the expectation would be for relocation costs in the Alps to be greater, as carnivore reintroductions could conflict with tourism, and the likelihood of individuals crossing borders increases international opposition (Rosen and Bath, 2009).

The assumption drawn from this data is that conducting MR for large carnivores could require a greater amount of financial investment, when it comes to investigating feasibility, ensuring public safety and gaining social acceptance, compared to a less hazardous species. Feasibility assessments are likely to require a lengthy engagement with the public, to ensure MR do not conflict with local interests, and make certain MR is accepted by local stakeholders and communities (Clark *et al*, 2001). This is essential in guaranteeing the long-term persistence of relocated populations, but could increase costs as a result.

The second highest expenditure was for translocating ten African elephants. Costs for this project are based on estimates and a majority of the outlay is for equipment totalling $100,000 needed for feasibility and background studies prior to relocation. By contrast, the 2009 translocation of sixty elephants compiled from an actual budget, shows there was no expenditure during this phase, due to the release site being pre-determined. This raises the question of the accuracy and usefulness of estimated data; despite being based on expert knowledge, it is open to conjecture to what extent estimates may differ from the actual costs. This issue is subject to statistical evaluations in a later chapter.

Of the fifty one relocations, forty two (82%) spent less than $1,000 per individual during this phase, and twenty eight (55%) spent less than $100 per individual. Current relocations may not necessarily incur high financial costs during this phase, as they are conducted within the historical range of a species. Release sites can be identified using species distribution maps and habitat prerequisites, and public acceptance is unlikely to be an issue for non-hazardous species, as existing relocation operations generally release species within their native ranges. However, the MR of species outside historical ranges will undoubtedly require a greater degree of financial investment into background research, compared to current relocation operations, it is imperative to assess a projects feasibility, investigate potential impacts MR could have in recipient ecosystems, and consider societal tolerance (Hunter, 2007; McLachlan *et al*, 2007; Hoegh-Guldberg *et al*, 2008; Richardson *et al*, 2009).

4.4 Step 3: Selection of Release Site

The Brown bear reintroduction also encountered the largest financial outlay during this phase. A total of $92,332 was spent on reports, PR literature, meetings and consultations, and $10,000 on staff wages; the cost per individual totals $10,233. The American manatee reintroduction incurred the second highest cost, with $5000 alone spent on meetings and consultations. Managers ensured the release site was occupied by wild Manatee populations, enabling the reintroduced juvenile to interact with wild conspecifics (Morales, 2010). Of the fifty one relocations, thirty nine (76%) spent less than $100 per individual during this phase, and twenty one (41%) spent nothing on release site selection. The majority of relocations (78%) took place in NR or NP where release sites were predetermined prior to project commencement. The costs incurred during this phase predominantly related to salaries, meetings and consultations.

The amount of disturbance MR could potentially cause, if recipient ecosystems are not appropriately selected, demonstrates the need for careful consideration to this phase prior to commencement (Hunter, 2007). MR may be deemed more acceptable in degraded areas, that would benefit from ecological restoration, rather than in protected areas containing unique biota (Hunter, 2007). MR release sites are unlikely to be predetermined, and selecting an appropriate release site for MR can pose certain difficulties, requiring greater financial investment than the expenditure faced by current relocations.

4.5 Step 4: Preparation of Release Site

The projects that incurred the greatest financial outlay during this phase were two of the African wild dog reintroductions. The cost per individual for installing new perimeter fences to contain animals within the release site was $98,984 ($890,862 in total) within a 75,000 hectare NR, and $57,840 ($636,243 in total) within a 55,000 hectare NP. The American manatee reintroduction spent $33,000 preparing the soft release facility for a single juvenile. The Mountain bongo project spent $26,388 per individual, installing fences in a 485 hectare release site, in total $475,000 was spent on installing, maintaining and electrifying perimeter fences.

Small mammal relocations were also subject to high costs during this phase. The Numbat reintroduction spent $16,535 per individual ($347,248 in total). The Greater stick-nest rat reintroduction in 2000 cost $5,495 per individual ($120,900 in total), and was spent eradicating invasive species from the release site that would have predated reintroduced populations. Despite this substantial outlay the stick-nest rat reintroduction was unsuccessful in establishing a population, although it remained unclear what factors caused this. Twelve relocations (23%) spent $1000 - $5000 per individual, and twenty seven relocations (53%) spent less than $500 per individual. Projects incurring substantially lower costs appeared to require little or no habitat modification or management prior to release. Most of the costs incurred during this phase related to staff wages, equipment and habitat modification and restoration.

It may be essential for MR programmes to confine animals within a release site, to ensure they remain within the novel habitat during acclimatisation, and to prevent individuals dispersing towards their previous bioclimatic ranges. MR of large carnivores could also require animals to be confined within a release site for public safety reasons. What the dataset appears to demonstrate, is projects that made certain preparations within the release site to accommodate relocated species, such as installing perimeters or removing invasive species, experience substantially increased costs. How much site preparation will be required for specific MR may be open to speculation, but certain species could require substantial preparation prior to release.

If MR is to mimic natural dispersal, as suggested by Vitt *et al* (2009), and have a low impact on ecosystems (Davidson and Simkanin, 2008; Mueller and Hellman, 2008) pre-release site preparation may not be necessary in all cases, thus economic costs could be lower than expected. However, when considering Hunter's (2007) suggestion that MR could form part of an integrated ecological restoration project, then site preparation would become fundamental, and the associated costs could be substantial.

If MR is undertaken with smaller mammals, and species vulnerable to predation, it may be harder to locate release sites with low predation risks. Furthermore, predators which are native to the recipient ecosystem may need to be managed, in order to facilitate the MR of novel species, posing an ethical debate, which is beyond the scope of this thesis. A study by Martins *et al* (2006) showed that costs for controlling invasive mammals on islands can run into hundreds of thousands of dollars, and eradication costs increase with land area. Hunter (2007) advocates that MR are likely to be more prevalent in mainland ecosystems, it is questionable how feasible it would be to control predators that could impact on relocated populations in large mainland areas.

4.6 Step 5: Transporting Individuals to Release Site

Staff salaries, husbandry during transportation, vehicle and fuel costs, and equipment needed for animal transportation comprise the majority of the financial costs associated to this step. Relocations requiring animals to be transported internationally met with the greatest cost during this phase. Transport costs for the Mountain bongo reintroduction amounted to $18,833 per individual ($339,000 in total), and the Dorcas gazelle reintroduction totalled $6,191 per individual ($160,971 in total). The Scimitar-horned Oryx translocation also incurred a large financial outlay of $4,707 per individual ($37,660 in total), this was due to animals having to be transported to three separate release sites. Of the remaining relocations, seven (13%) spent between $1,125 and $3,327 per individual, and forty-one (80%) between $0 and $960 on transportation.

The distance individuals are transported during MR is likely to vary between species, and may be largely dependent on predicted shifts in a species bioclimatic envelope. The Torreya Guardians have controversially been relocating the Florida Torreya conifer over a distance of 643 Km northward of its native range (Marinelli, 2010). Willis *et al* (2009) have conducted experimental MR with two British butterflies; individuals were moved up to 65 Km beyond historical northern range margins. It is expected that MR will be undertaken within the broad geographic region of a species (Hoegh-Guldberg *et al*, 2008). For many species, a relocation of 500 Km, or even 1000 Km, may be required, in order to locate suitable climatic conditions (Svenning *et al*, 2009). Financial outlay substantially increases when species are transported internationally to a release site. If certain species begin succumbing to CC, it will become impracticable to source these individuals from wild populations, and managers may be faced with no alternative option than to use captive bred animals, sourced from international ex-situ breeding programmes. This will undoubtedly increase the travel cost, and financial limitations could lead to MR becoming economically unviable for species requiring long distance transportation.

4.7 Step 6: Release Strategy

The American manatee reintroduction required the most expensive release strategy, and spent a total of $45,500 during the 8 month soft release of one male calf. Most of the financial outlay was attributed to staff wages, husbandry, veterinary care, release facility maintenance and education programmes. The 1999 African wild dog reintroduction also spent a considerable amount on husbandry during a soft release ($11,010 per individual), unfortunately timescales for this release were not provided by the project manager. The Brown bear reintroduction spent $27,833 per individual during the release phase; this was attributed solely to the salaries of researchers, consultants and rangers over the 8 year project cycle. Although some projects incurred high costs, 35 relocations (69%) spent less than $50 per animal released.

Some MR could possibly use soft release strategies, similar to the American manatee and African wild dog reintroductions, allowing animals to recover from relocation stress, and acclimatise to release sites; particularly important when releasing animals into novel ecosystems outside their native range. Animals placed directly into ecosystems they are unfamiliar with may cause distress and disorientation, potentially reducing the long-term persistence of relocated populations (Fa *et al*, in press). It is imperative animals that pose a health risk to the recipient ecosystem are not released. Keeping animals in quarantine at the release site means they can be monitored and the necessary veterinary care can be administered as required (Armstrong and Seddon, 2007). The direct release of animals into recipient ecosystems may prove a less expensive option, but not taking the necessary steps to ensure species persist in their new habitat, and individuals are healthy prior to release, may lead to valuable funds being wasted. Therefore it is reasonable to expect that MR will adopt soft release strategies, ensuring species relocations are managed appropriately. This can potentially increase the costs associated with this phase, in comparison to current relocation practices.

4.8 Step 7: Post Release Activities

Projects that incurred the greatest financial outlay during the post release phase included three of the African wild dog reintroductions, these projects spent between $66,184 and $78,284 per individual. Costs were attributed to salaries, husbandry (post release supplementary feeding), veterinary care and reducing human/wildlife conflict. The reintroduction of Otters in the Netherlands is ongoing; running for a period of eight years to date. The post release stage of this operation is the fourth most expensive, totalling $65,349 per individual, $1,817,142 in total was spent on reports, consultations, data analysis and salaries, $60,000 was spent hiring planes needed for telemetry monitoring and $18,000 on tracking equipment. The Brown bear reintroduction also incurred high financial costs; during the eight year project a total of $127,000 was spent on transport and equipment and $278,333 on staff salaries, equating $44,758 per individual. During this phase twenty one (41%) relocations spent between $8,000 and $1,000 per individual, and twenty two (43%) spent less than $1000. The Chacoan peccary reintroduction was the only project which did not incur costs during this phase. Acquiring animals, and conducting feasibility studies and genetic analysis were the only areas which required financial investment. Managers of this project were able to keep the monetary outlay to a minimum by involving local communities in release and monitoring activities (Benirschke, 2010).

If MR is considered an appropriate measure only once a species is endangered, which often indicates populations have been pushed to a critically low level, it will be difficult to sustainably harvest the numbers needed for relocation from reduced wild populations

(Armstrong and Seddon, 2008). Relocating small groups of animals poses various problems (e.g. Wolf *et al*, 1996, 1998;), such as locating individuals during post-release monitoring phases (Beck *et al*, 1994). Projects observing populations at low densities are likely to need more staff for this purpose, or may have to invest in expensive monitoring equipment. Both factors can impact upon the amount a project is required to invest in post-release monitoring.

The financial investment required for undertaking post release activities is likely to be dictated by the focal species. Species requiring extensive veterinary care, post release supplementary feeding, and those that are difficult to monitor, or pose potential safety hazards, are likely to be the most expensive to relocate. Involving local communities could present a cost effective way of conducting essential post release activities; this is reliant upon people and their willingness to engage, and managers will also need to ensure that tasks are completed effectively. This approach has the benefit of engaging communities, raising public awareness of relocation issues and potentially increasing societal acceptance of MR.

If MR does take place it will be imperative that extensive long-term monitoring programmes are incorporated into the planning process. This is essential in order to ensure that ecosystems are not adversely disrupted, that relocated populations remain healthy, and provide tangible evidence that MR is a viable management option that meets objectives. In this respect MR could be comparable, in terms of financial outlay, to relocations that have conducted long-term post release monitoring activities in the past.

4.9 Step 8: Reporting and Assessment

During reporting and assessment stages the Brown bear reintroduction incurred the greatest financial outlay. Over the course of the eight year project $278,333 was spent on salaries and $92,332 on publishing reports and conducting meetings, amounting to $37,066 per individual. The dataset revealed that costs for this phase varied, even between projects relocating the same species. Of the six African wild dog reintroductions one spent $18,505 and the other $2,040 on monitoring and assessment per individual, while the remaining four spent nothing on this phase. Bighorn Sheep relocations demonstrated a similar trend; the 2004 reintroduction in South Dakota spent $3,521 per individual, whereas the three translocations in New Mexico spent less than $100 per individual. During this phase twelve (23%) projects spent between $1,000 and $100, and thirty (59%) spent less than $100 per individual. It is apparent from the dataset that only twenty two (43%) of the relocations invested funds into reporting and assessing post-release outcomes.

This indicates an urgent need for enhanced post release reporting and improved documentation of past relocations. It is essential that managers publicise project outcomes, positive or negative, to ensure that future relocations can improve upon the methods adopted by previous projects (Fischer and Lindenmayer, 2000). If MR guidelines are to be developed, and project efficiencies are to increase with experience, it is imperative that managers communicate their experiences, and invest time and funds on appropriate reporting and assessments.

5. Statistical Analysis

A readout of the statistical analysis can be found in Appendix 2. All numerical data has been transformed into Log10 values. An analysis of variance (AOV) was conducted using R.2.11.1 to determine whether any of the explanatory variables had a statistically significant explanatory power, providing an indication of potential costs for MR. The first step AOV was conducted using body mass (P=0.07), conservation status (P=0.05303), GDP (P=0.04244), and region (P=0.62749) as explanatory variables; data source (P=0.69539) was also included within the maximal model to ensure no significant differences existed between estimated and actual costs. A backward stepwise regression was performed on the maximal model, to indentify a preferred model containing fewer significant parameters. Akaike's information criterion (AIC) was used to select each variable in turn. Body mass (AIC= -21.529), conservation status (AIC=-21.108), and GDP (AIC=-25.030) were identified as the most significant explanatory variables. Region and data source were removed from the model, as these variables had the least significant explanatory power.

Accuracy of the stepwise regression was verified by updating the model and removing the most significant variable, which was GDP as this had the lowest AIC value. A second step AOV compared the two preferred models with and without GDP included. This AOV showed a significant difference between the models (P=0.04244), which confirmed the removal of GDP made the preferred model significantly less reliable. The third step AOV was conducted using body mass (P=0.06799), conservation status (P=0.04462), and GDP (P=0.09317); GDP proved to be the least significant variable. The fourth step showed that body mass (P=0.07385) was less significant than conservation status (P=0.05285). The fifth and final step confirmed that conservation status alone (P=0.07756) was greater than the required significance level. Therefore, the minimal adequate model is a null model, as none of the variables had statistically significant explanatory power in predicting variation between relocation costs.

6. General Comments on the Dataset

Statistical analysis of the dataset revealed no correlation between cost per individual released and GDP (Figure 6.1). It may be reasonable to expect project costs to be significantly lower in countries with a lower GDP (e.g. Balmford *et al*, 2003). This did not prove to be the case with the relocations incorporated in this study; potentially because costs for equipment and consumables do not vary in relation to GDP, or, relocations are expensive operations regardless of the country where they are carried out.

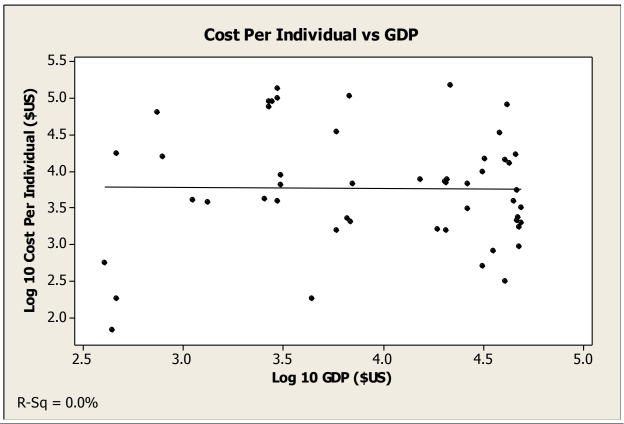

Figure 6.1: Cost per individual released in relation to the GDP of the country where the relocation project was conducted.

The total cost of each relocation showed no correlation with GDP (R-Sq = 0.6%), and there was no correlation between the number of individuals released and GDP (R-Sq = 1.3%), indicating that cost per individual was not skewed by disproportionate numbers of individuals being released within regions with a low or high GDP. What the dataset does indicate, is that the number of relocations undertaken increases with GDP, as more data points are clustered to the right of Figure 6.1. This reiterates observations made by Balmford *et al* (2003), that the majority of conservation expenditure is in developed countries.

Analysis showed no correlation between cost per individual and body mass (Figure 6.2**).** Despite the large amount of spread within the data, there is a negligible positive correlation, and it could be tenuously implied that body mass may have some connection with increased relocation costs. However, incorporating more data, for a greater variety of species, provides the only possibility of increasing the explanatory power of this model.

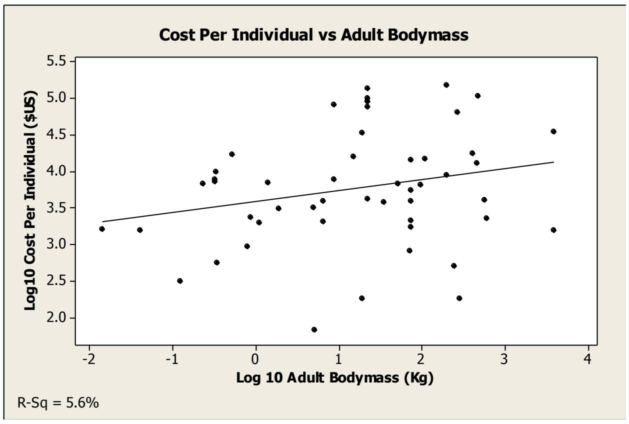

Figure 6.2: Cost per individual released in relation to species adult body mass.

Conservation status does not significantly correlate with financial expenditure, although costs do appear to rise from species of least concern to endangered species categories (**Figure 6.3**). The critically endangered category contains data for five species relocations, and the extinct in the wild category contains data for only a single relocation, a more comprehensive dataset could possibly give a better indication as to whether a species conservation status has a noticeable effect on the financial cost of relocation projects.

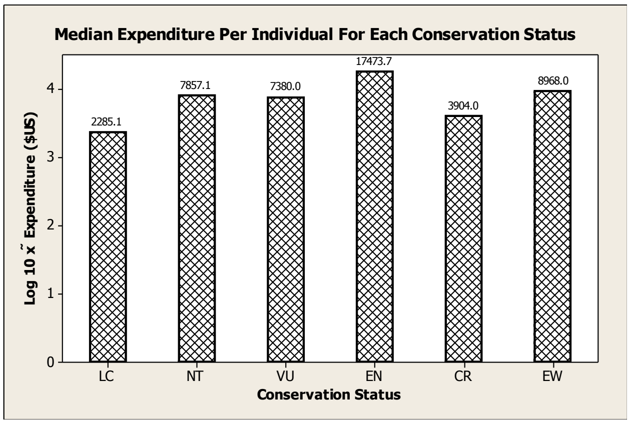

Figure 6.3: Median ($\tilde{x}$) expenditure per individual within each IUCN Red List classification. (LC) Least Concern, (NT) Near Threatened, (VU) Vulnerable, (EN) Endangered, (CR) Critically Endangered, (EW) Extinct in the Wild.

7. Problems with the Data and Project Limitations

Management approaches adopted by each relocation varied, making financial comparisons harder to accomplish than originally anticipated. Cost per individual gave an indication of expenditure needed to relocate a given species, and enabled comparisons to be made between projects; with limitations. Cost per individual increases with project duration; invariably impacting upon the overall project cost (Figure 7.1).

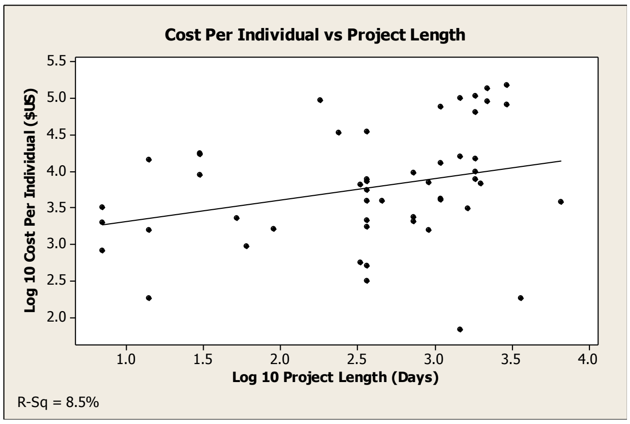

Figure 7.1: Cost per individual released in relation to the length of time needed to conduct the relocation.

To improve the accuracy of future studies, participants should indicate how long each project phase took to complete, so cost per individual, per unit of time, per phase, could be compared, eliminating the bias generated by ambiguity of phase lengths. Cost per individual decreases with the number of conspecifics released, indicating that releasing species in a large group reduces costs (Figure 7.2).

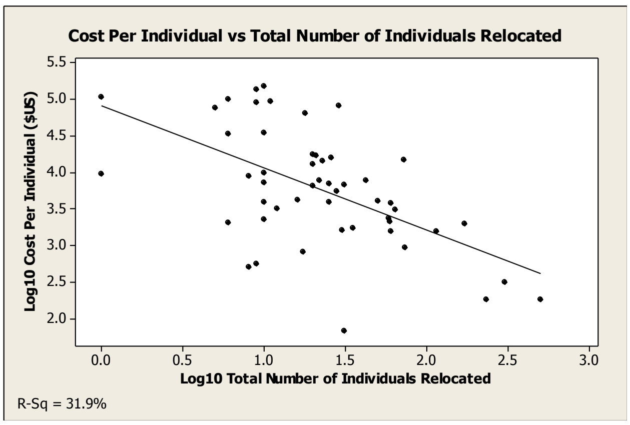

Figure 7.2: Cost per individual in relation to the number of individuals relocated. Cost per individual decreases considerably with founder group size. This bias within the dataset may falsely indicate that species released in large founder groups are more cost-effective. In reality project costs have been divided over a greater number of individuals within the founder group.

The majority of relocations incorporated in this study, successfully established populations within the budget indicated. The recorded periods of time for the completion of current relocations can only be used as an indicative measure of timescales needed to complete MR, which is largely untested and likely to take a protracted period of time in comparison; an important consideration if duration has a bearing on project cost. Current relocations move species within historical ranges where conspecifics are often present, whereas MR may transfer species to sites where no populations exist. Tudge (1997) suggests that no wild population is considered secure unless it contains around 500 individuals, in all likelihood MR will be required to move greater numbers of individuals than current relocations, potentially inflating project costs. Budgets published within this dataset fail to account for costs associated with long-term captive breeding needed to maintain populations ex-situ prior to release, therefore concealing the cost of acquiring a species for relocation. As climatic zones shift, novel ecosystems may arise outside protected areas (Burns et al, 2003), before MR is undertaken it may be necessary to purchase suitable release sites, but potential land purchase costs are not factored into the dataset; this is of considerable importance when evaluating potential costs of MR.

According to O'Brien *et al* (1999) there is up to 4800 extant mammal taxa, the dataset herein represents 38 (0.79%) species, and is unlikely to be representative of mammals in general. Mammals are also likely to be the most expensive species to relocate, compared to reptiles, amphibians and invertebrates (Willis *et al*, 2009). Given that mammals account for only 0.25% of global species (Rahbek, 1993), future investigations into relocation costs might consider incorporating other groups that may be better suited, or more cost effective, MR candidates. Time constraints due to project deadlines were a limiting factor. Providing participants with more time to gather information, may facilitate a better response in future studies. Many participants experienced problems gathering the required information; numerous participants communicated their willingness to take part; but admitted gathering data would be too time consuming or that supplying accurate data for projects running over long periods of time would not be feasible.

Some costs are hard to attribute exclusively to relocation activities, because resources and personnel serve multiple purposes; costs are concealed within broader management budgets. Many relocations are multi-organisational operations, conducted by various project partners, making the calculation of costs incurred by individual institutions complex. Furthermore,

certain organisations failed to maintain accurate accounts for each relocation; managers had a tendency not to view them in their individual capacity, but instead as part of general activities, as a result costs for specific projects are not documented.
This study offers a valuable insight into costs associated with mammal relocations, but inconsistencies between projects suggests that caution should be exercised when using the expenditure of previous relocations to forecast future project costs. The changeability of social, logistic, ecological and abiotic factors, and the unpredictability of environmental change, suggests the only thing practitioners can be certain of, is the degree of uncertainty (Saunder *et al*, 2007). Using past conditions as benchmarks for the future may be unfavourable, or even counterproductive in a time of global change (Cole *et al*, 2008), but investigating previous relocations, and examining reintroduction biology literature, can offer an insight of the issues managers and scientists may face, and should be aware of, when assessing the viability of MR.

8. Success and Failure of Previous Relocations

A number of previous studies suggest that many relocations are unsuccessful (Griffith *et al*, 1989; Kleiman, 1989; Dodd and Seigel, 1991; Wolf *et al*, 1996), can be extremely expensive (Kleiman *et al*, 1991; Lindberg, 1992; Rahbek, 1993), and instigated further research into factors influencing relocation success. Beck *et al* (1994), Wolf *et al* (1996) and Fischer and Lindenmayer (2000) emphasize that relocations using wild sourced populations are generally more successful, this shall be discussed further in a subsequent chapter. Griffith *et al* (1989) suggest that a relocated populations persistence is increasingly likely when the number of founders is large, population increase is high, fluctuation in rate of increase is low, there is little impact from competitors, and environmental variation is minimal. Omnivorous food habits, high genetic diversity among founders, and small body mass may also improve chances of persistence (Wolf *et al*, 1998). Griffith *et al* (1989) advocate that relocations of widespread species were more likely to be successful, than those conducted with endangered or sensitive species, and that species which breed early and bear a greater number of offspring are more likely to persist. Conversely, studies by Wolf *et al* (1996; 1998) found that reproductive potential does not necessarily influence a populations persistence.

Relocations that release a greater number of individuals are generally more successful (Griffith *et al*, 1989), as environmental stochasticity and adverse demographic effects more

commonly affect small populations (Wolf *et al*, 1998). However, it is also possible to establish populations with a low number of founders (e.g. Taylor *et al*, 2005). Small founder groups are released when it is perceived as highly probable that relocations will fail, or are poorly resourced; creating a bias toward the success of large groups (Armstrong and Seddon, 2008). Determining the minimum number of individuals required to guarantee population persistence will depend on the circumstance of each relocation; using population viability software can prove useful, by providing minimum population estimates (Wolf *et al*, 1998). If there are insufficient individuals available to conduct a single relocation, managers may consider releasing the minimum number of individuals over a number of years to increase the likelihood of persistence under constrictive circumstances (Griffith *et al*, 1989). Undertaking relocations should be considered before they become a last resort for managers; once a species reaches a low population density, obtaining a founder group becomes more difficult; compounded by the potential failure to successfully establish a new population, the overall effects of which could push a species closer to extinction (Griffith *et al*, 1989).
Griffith *et al* (1989) propose that the best candidates for relocation are species with expanding populations. Presenting a convincing argument for implementing MR with a species whose population is increasing, may prove a challenging paradox.

Simply releasing large numbers of individuals will not guarantee success, a sufficient quantity of suitable habitat for the relocated species is a prerequisite, and is generally cited as the most common factor influencing relocation outcomes (Wolf *et al*, 1996). Armstrong and McLean (1995) advocate that relocation procedures should incorporate thorough assessments of habitat quality prior to release. Identification and preservation of suitable habitat may require a combined ecosystem and species approach, requiring managers to gather ecological information on factors that impact habitat quality, such as: species life history traits, ecological interactions and minimum habitat requirements (Griffith *et al*, 1989). One factor that could pose problems for MR is that individuals released at the periphery, rather than the core, of their historical range are less likely to persist (Wolf *et al*, 1998). However, Griffith *et al* (1989) indicate that species relocated into areas without competitors are more likely to establish. Therefore, selecting a release site based upon competitor absence, counteracts the pressures exerted on a species moved out of its core distribution.

Non-biological and technical factors also contribute to relocation success. Reading *et al* (1997) highlight that valuational and organisational factors, such as public relations (PR), education programmes and specialised working groups, can influence relocation effectiveness, and suggest that inadequate staff training constrict practitioners abilities to recognise social science variables. Reading *et al* (1997) show that relocations adopting PR and education strategies, using wild-caught animals, and re-establishing extirpated populations, received the highest local public support. Relocations considered failures (i.e. released animals failed to establish) had significantly lower local support. Simply undertaking PR and education programmes won't guarantee community collaboration, or success, but they should be used to encourage local backing, or at least dissuade opposition (Reading *et al*, 1997).

Reading *et al* (1997) show that organisations employing a high number of staff to conduct relocations generally suffer an increased number of conflicts than those with fewer people. In addition, Clark *et al*, (1989 and 1994), Clark and Westrum (1989) advise that specialist teams should be staffed with well trained experts in species recovery, rather than simply including members of certain agencies or interested parties. MR may require separate teams to focus on biological or technical approaches, as well as socioeconomic and political consideration.

This research highlights considerable variations between projects; managers should be aware of the implications of applying generalisations across a range of taxa when considering what contributes to relocation success. The general assumption is that a viable self-sustaining population indicates relocation objectives have been achieved, however, success is contemporaneous; as new threats can arise at any given time (Seddon, 1999; Fischer and Lindenmayer, 2000). Individual managers will be responsible for defining what constitutes a success, for their specific MR programme.

9. Challenges and Lessons: Using Previous Relocations to Understand Managed Relocation

Existing relocation literature is a valuable source of information, yet it mainly consists of retrospective analysis, driven by available data, rather than deriving from questions posed *a priori* (Armstrong and Seddon, 2008). Research historically has been ad-hoc, rather than an organised attempt at attaining knowledge for the improvement of species relocations, in order to guide future studies on this subject. There has been a rise in modelling approaches that inform future relocations, yet there are few examples of experimental tests of explicit hypotheses (Seddon *et al*, 2007). Walters and Hollings (1990), Nichols (1991) and Roush

(1995), highlight that using model species in relocation experiments can often limit the range of testable hypotheses. Research questions need to be identified, and appropriate methods used, in an attempt to provide answers to the MR debate, rather than focusing on limited questions, that lend themselves to precise scientific assessments (Armstrong and Seddon, 2008). An experimental approach that evaluates relocation management will help to prevent untested ideas entering into MR practice (Seddon *et al*, 2007).

Experimental management approaches generally lack controls, replicates and adequate monitoring, making it difficult to assess which variables contributed to a positive outcome (Seddon *et al*, 2007). Fischer and Lindenmayer (2000) suggest that only successful relocations tend to be documented, making it harder to ascertain what has caused previous relocations to fail (Seddon *et al*, 2007). In order to develop the science behind MR, managers might consider following Williams's (1997) recommendation, which suggests that undertakings should be guided by intuition, traditional convention and assumptions. Observations should be organised into coherent categories, obvious patterns explored, the causes of trends should be indentified and theoretical predictions tested. However, it may be difficult, and costly, to replicate and perform controlled experiments, to tackle the uncertainties of MR; especially when working with endangered species (Bennet and Adam, 2004; Armstrong and Seddon, 2008). In cases where experiments are not feasible, an alternative or complimentary approach could be adopted, such as focussing on management practices and relocation outcomes, based upon the available data relating to reproduction, survival and dispersal rates for focal species and ecosystems (Komers and Curman, 2000; Towns and Ferreira, 2001; Seddon *et al*, 2007; Armstrong and Seddon, 2008).

Game translocation could provide an ideal opportunity for managers to improve their understanding of MR requirements. In 2008 the South African Wildlife Translocation Association asked members to provide details of their activities; 11 responded declaring that 46,500 animals had been moved within that year (Currie, 2010). In line with suggestions put forward by Sarrazin and Barbault (1996), to mimic natural colonization, MR will need to ensure that individuals are not drastically altered by captivity or the relocation process, mimic the age and sex of natural dispersers, and that relocated population numbers should be likely to occur in nature.

10. Ex-situ Conservation: Improving or Undermining Managed Relocation?

Effective in-situ conservation programmes can be expensive, but the costs of ex-situ conservation, especially for large threatened mammals, will generally be much higher (Leader-Williams, 1990; Balmford *et al*, 1995). Balmford *et al* (2003) also showed that the costs for ex-situ conservation in three zoological parks in the U.K. and U.S.A. cost between \$6 million and \$160 million per Km^2, per year, implying that zoos could increase their cost-effective contributions to conservation practices, by implementing more field based projects. Investing funds to investigate innovative ways of effectively protecting biodiversity in-situ, could prove more effective, than instigating additional captive breeding programmes to protect species from CC. The operational cost of ex-situ population recovery can run into millions of dollars, per species, per year (Derrickson and Snyder, 1992). Some of these costs can be met by institutional revenues, but this could lead to the preferential selection of crowd pleasing 'box office' species. Many endangered species are not aesthetically pleasing, and therefore present a limited potential to fund their ex-situ conservation through exhibition income. Financial backing from private and government sources is often limited, and is usually only granted to species or programmes that have substantial public appeal (Snyder *et al*, 1996).

As previously discussed, relocations are more likely to succeed when using wild sourced population (Beck *et al*, 1994: Wolf *et al*, 1996; Fischer and Lindenmayer, 2000). This is often because animals lose natural behaviours associated with wild fitness when they are kept in captivity; behavioural deficiencies have been observed in foraging and hunting, breeding and nesting, locomotory skills and social interactions (Snyder *et al*, 1996; Rabin, 2003, Stoinski, 2003; van Heezik and Ostrowski; Vickery and Mason, 2003). Furthermore, captive-born individuals can also lack immunities to diseases, viruses and parasites prevalent in wild conspecifics (Cunningham, 1996). Evolutionary processes do not cease when animals are kept in captivity; some species have shown major changes in morphology under captive conditions. If a species become increasingly adapted to life in ex-situ facilities, they can lose the necessary survival skills needed to persist in the wild, and may even exert deleterious genetic pressures on remnant wild populations (Philippart, 1995; Snyder *et al*, 1996). Maintaining species in captivity long-term to protect them from CC, or whilst practitioners decide upon the best course of action, could seriously limit the chances of populations being restored to the wild, highlighting the need for inventive in-situ conservation techniques to be developed.

To date only a small number of the world's taxa have been bred in captivity (Conway, 1986; Rahbek, 1993), and obtaining reliable survivorship and reproduction has proven difficult for numerous species. Identifying factors that limit breeding success can be complex and costly, for many endangered species, effective ex-situ management is still proving unsuccessful, despite years of experimentation (Snyder *et al*, 1996). Additionally, zoological institutions do not have the space to accommodate viable populations for all threatened species (Soule *et al*, 1986; Balmford *et al*, 1995). Of a possible 30 million species inhabiting the planet, a large proportion live in rapidly declining and fragmented tropical forests, therefore half could be broadly considered endangered, neither zoos nor other conservation facilities have the capabilities to protect such a vast number of species (Tudge, 1997).

Zoos are in the powerful position of helping to finance in-situ conservation, as well as raising public awareness and support for conservation initiatives (Conway, 1992; Balmford *et al*, 1995). Species at the top of the food chain, whose behaviour is instinctive, or those being relocated into predator free environments, may well be suitable candidates for short-term captive breeding programmes, which could be used to enhance population numbers prior to relocation (Snyder *et al*, 1996). The 'ark' paradigm that envisages numerous species being maintained in captivity for hundreds of years is unattainable, and undesirable as long periods of time in captivity can decrease the possibility of a successful reintroduction. Ex-situ conservation should not be deemed a long-term solution, and should always be integrated with efforts to re-establish populations in the wild (Snyder *et al*, 1996).

The existence of populations in captivity may give the false impression that a species is protected, undermining efforts to conserve wild populations, and combat the threats driving them to extinction. Maintaining species in artificial ecosystems may pose less of a risk than MR, yet if captivity reduces the chances of restoring individuals back to free-living populations, ex-situ conservation should be considered the last resort (Hunter, 2007). Managers must ensure that captive breeding programmes, if utilised as part of MR, are employed to improve the chance of relocation success, rather than using them to delay exploration into novel approaches of in-situ conservation.

11. **Predicting the Future**

Modelling strategies adopted in order to predict the impact that CC may have on biodiversity often focus on the 'bioclimatic envelope' of a species (Bakkenes *et al*, 2002; Berry *et al*, 2002; Pearson *et al*, 2002). These techniques correlate current species distributions with climate variables, or their physiological response to CC (Franklin, 1995; Mack, 1996; Guisan and Zimmerman, 2000). Once a species climate envelope is established, future climate scenarios can be applied to identify potential species redistribution. Bioclimatic modelling in its purist form fails to consider the effects that other environmental factors, such as soil type and topography, have on species distributions, although advances in this approach allow for environmental factors to be factored, along with climate envelope variables (Pearson and Dawson, 2003; Gelfand *et al*, 2005; Guisan and Thuiller, 2005; Elith *et al*, 2006; Wright *et al*, 2006).

A correlative approach incorporates the knowledge pertaining to the ecology of a species, yet it remains unclear how this will change under future conditions, especially in terms of interspecies relationships. Furthermore, a correlative approach is based upon species observations as we perceive them today, current species distributions may not be in equilibrium with the present climate (Pearson and Dawson, 2003). Many endangered species are data deficient, therefore it is unlikely there will be enough information available to accurately model their potential distribution. Correlating recent climate with species distribution may fail to fully identify a species potential climatic range, therefore bioclimatic modelling should always be interpreted with a degree of caution (Pearson *et al*, 2002). The experimental MR of Butterflies by Willis *et al* (2009) demonstrates how bioclimatic modelling can be used effectively, by evaluating the multitude of interacting factors, assessing species potential dispersal abilities, and acknowledging limitations and uncertainties of bioclimatic modelling, all of these factors should be considered, to ensure realistic applications of CC impact simulations are achievable (Heikkinen, 2008).

Climate driven evolutionary change should be considered as part of MR assessments. It is expected that evolutionary change takes place over long periods, and species tolerance range remains constant as it shifts its geographical distribution (Pearson and Dawson, 2003). However, Woodward and Rochefort (1991), Davis and Shaw (2001) and Thomas *et al* (2001) have shown that climate induced range shifts select against phenotypes that are poorly adapted to shifting climatic conditions, the implications of which need to be considered to ensure that MR does not disrupt natural evolutionary adaption processes, and it must not be assumed that all species will show similar adaptive responses (Pearson and Dawson, 2003).

12. <u>Managing Novel Ecosystems</u>

Management policies and legislation applied to protected areas often indicate that preservation, protection, and unimpairment is achievable by maintaining 'naturalness'; this is a guiding concept for environmental stewardship that has remained largely unchallenged (Cole *et al*, 2008). White and Bratton (1980) maintain that natural states are unattainable because environmental systems are dynamic. Ecosystems are unique in space and time so arguably it is impractical, or even undesirable, to maintain them in their current state perpetually; existing systems are likely to become unstable in the future due to CC (Gillson and Willis, 2004; Harris *et al*, 2004). Countless managers have used past environmental conditions as a benchmark for future condition. This has lead to a 'purist' concept often being applied to conservation management, despite the fact that there is evidence to suggest that in many cases ecosystems need to be artificially manipulated in order to maintain a 'natural' state (Cole *et al*, 2008). Cole and Landres (1996) expound that even remote wilderness ecosystems have been, and will be, affected by human activities. Scientist must start to challenge whether the significance of 'naturalness' is still an adequate guiding concept in a era where the realm of human interference has gone global, and the number of advancements and available knowledge on this subject is ever increasing (Cole *et al*, 2008).

Ridder (2007) states that a choice has to be made between protecting biodiversity and respecting natures autonomy. Future climates have no analogue; therefore conservation practitioners may be faced with the challenge of redefining 'naturalness' to ensure ecosystems can endure the ongoing impacts of human activities and future CC (Cole *et al*, 2008). In 1988 Canada's National Parks Act adopted 'ecological integrity' as their management objective. Ecological integrity is defined as a condition that represents the characteristic state of a natural region that is likely to persist, measured by abiotic components, native species, and biological communities, their rates of change, and the supporting processes (Cole *et al*, 2008). If conserving biodiversity is considered paramount, protected areas seeking to maintain ecological integrity might have to consider MR in order to help maintain community structure, even if it means that the ecosystem composition is no longer considered ‘natural’ by purist definition.

An ecosystems ‘resilience’ to extreme and unpredictable environmental change has also emerged as a management guidance concept (Cole *et al*, 2008). Holling (1973) defines ‘resilience’ as the capacity of a system to cope with change, and persist without undergoing a loss of fundamental character. Resilience theory states that initially withstanding gradual environmental changes means the affect of future impacts are magnified, and that conditions should not be preserved to the detriment of a systems resilience in the future (Cole *et al*, 2008). Managing ecosystem ‘resilience’ emphasises the importance of retaining fundamental system functions, in preference of preserving specific species in-situ, implying that the landscapes of today may have to be altered to ensure core ecological processes remain resilient to environmental change. When defining the desired outcomes of future management strategies, practitioners will need to consider the value of conserving biodiversity, preserving ecosystem integrity, aesthetics and nostalgia, as well as ensuring areas persist where nature can remain undisturbed, to run its own course (Cole *et al*, 2008). Under future climate scenarios MR may play some part in maintaining an ecosystems integrity, increase its resilience against CC, conserve biodiversity and maintain the aesthetics of landscapes.

13. How Could Managed Relocation be Approached?

MR programmes will need to adopt an adaptive management approach. Activities will be, in some respects, experimental and designed primarily as a means to provide information (Holling, 1978). MR strategies should be continually reviewed, as adopting predetermined procedures is not fundamentally conducive with MR (Lee, 1999). Ecosystems could be manipulated over time and/or space to determine which management prescriptions increase the likelihood of MR successfully establishing populations within recipient regions. It is important that findings are recorded, so that subsequent MR programmes can adopt and develop strategies, using techniques that have proved successful in the past (Seddon *et al*, 2007). Approaches to monitoring MR should planned in detail prior to project initiation, to ensure that potential issues are indentified and addressed, that effective methods of study are implemented in the first instance, and to ensure that research relating to MR practices is constantly updated and improved. Project monitoring and evaluation should not focus solely on the focal species, but also on the broader ecosystem of the release site, and meta-population dynamics (Armstrong and Seddon, 2008).

Utilising prior knowledge, to clarify uncertainties surrounding potential MR risks and benefits, could enable practitioners to devise management plans that consider, and address, the areas where greatest uncertainty exists. Where possible, MR should be replicated and controlled, so that the effects population foundation has on particular biological traits, or ecosystem function, can be monitored (Sarrazin and Barbault, 1996). Relocated populations that persist are likely to correspond with dynamic processes that occur within the recipient ecosystem (Sarrazin and Barbault, 1996). It will be necessary to ascertain demographic parameters, as well as model various environmental scenarios, so that population growth rate and variance can be monitored, which will enable the success of MR to be examined. Population modelling should be integrated into all MR evaluations to facilitate long-term population viability assessments. This will require collaboration between managers and scientists, to develop and refine modelling processes that are needed to explore and adapt suitable MR protocols. Habitat conditions vital for the persistence of a relocated population, such as resources, existing parasites and predators, should be considered prior to relocation. It is essential that methods used to assess these factors capture data that is relevant to species being relocated, especially when rapidly acquired data, such as geographical information system layers, is used to inform decision making (Armstrong and Seddon, 2008).

Genetic makeup of founder populations is likely to be a key factor affecting the outcome of MR programmes. A relocation is likely to fail if founder groups are inbred, or of inapt provenance (i.e. not suitably adapted to conditions within the recipient region). MR may only be able to accommodate moving a small proportion of a population, thus the founder population will possess reduced genetic diversity, which overtime could result in immunocompetence or inbreeding depressions, a problem that will need to be identified and addressed appropriately (Armstrong and Seddon, 2008). To counteract these issues, periodical introductions of new individuals into the relocated population may be necessary. However, it crucial that this does not have negative genetic consequences, such as inhibiting local adaption (Armstrong and Seddon, 2008). The impacts of removing individuals from source populations must also be taken into account. The number of individuals that can be sustainably ‘harvested’ from these populations must be carefully scrutinised, using population modelling, and will require accurate projections to be based on sound understandings of population regulatory mechanisms (Dimond and Armstrong, 2007).

Community interactions, such as competition, trophic associations and mutualism, may make it necessary to conduct multispecies MR. Ecological interaction will have to be determined as MR, it may be necessary to include obligatory resources and mutualists in the relocation programme. To investigate the impacts this could have on MR success it is imperative that taxa whose distributions are determined by limiting species, such as herbivores whose ranges are governed by the presence of food plants, are identified, along with their obligatory resources, and the implications for relocating each species should be assessed (McLachlan *et al*, 2007). The order in which species are relocated should be taken into account, especially when considering relocating those at different trophic levels to the same site. The functional responses between relocated species could influence their ability to coexist, alter community structure, or modify the anticipated impact a species has within the recipient ecosystem; all of which will affect the overall outcome of MR (Armstrong and Seddon, 2008).

Studies by Svenning *et al* (2009) suggest that the degree to which a recipient community is saturated with species may be a crucial consideration, when deciding the optimal allocation of individuals amongst release sites. However, there is still some debate over whether communities can actually become saturated (Cornell, 1999; Hillebrand, 2005; Ricklefs, 2006; Stohlgren, 2008). The relationship between diversity and ecosystem stability (e.g. Hooper *et al*, 2005) indicates that species rich ecosystems may be less likely to be disrupted by MR, compared to species poor ecosystems; although Hunter (2007) believes that this judgement should remain open to debate. However, Svenning *et al* (2009) propose that due to mounting evidence against community saturation, there is little reason to expect that MR will inevitably result in the loss of native species from the recipient ecosystems.

Selecting suitable release sites will undoubtedly require a considerable amount of planning. Hunter (2007) suggests that well connected sites, that have undergone major shifts in community composition in response to natural CC, or sites previously well connected that have become fragmented by human activities (which provides a barrier to dispersal should MR inadvertently have unacceptable effects) are preferable. A controversial, although interesting approach, is the potential of MR as a means of realigning ecosystems, to cope with environmental change, particularly systems that are already beyond the range of natural variability (Cole *et al*, 2008). Sites could be selected by assessing the amount of degradation that would occur under CC without active transformation, ecosystems at risk of becoming highly degraded, and would therefore benefit from active restoration, could be considered as potential MR release sites. Conducting MR could be used to restore ecosystem resilience, or develop novel ecosystems, suitable for new climatic regimes (Armstrong and Seddon, 2008; Cole *et al*, 2008). MR should be avoided within sites where ecosystems have evolved in isolation, containing unique endemic biota, and wilderness sites of particular importance, that function as 'controls' within a landscape of actively managed systems (Hunter, 2007; Cole *et al*, 2008).

Selecting candidate species suitable for MR offers a particular challenge, and is subject to intense debate. Distribution models have been identified as more accurate for species with broad distributions (Berry *et al*, 2002), than for species with narrow distributions (Gelfand *et al*, 2005; Guisan and Thuiller, 2005; Elith *et al*, 2006; Wright *et al*, 2006). Extinction risks for narrowly distributed taxa could be difficult to predict, leading to over or underestimation of a species potential risk and/or dispersal capabilities (McLachlan *et al*, 2007). Long-distance dispersal is another important consideration, but this is hard to characterise as small errors in estimations can result in considerable miscalculations of natural range shifts (Clark et al, 2003; Trakhtenbrot, 2005). MR programmes will be required to select source populations judiciously, to ensure founder groups contain genotypes most likely to persist within the recipient region. Individuals from the northern periphery of a species range may be selected, as they are the most likely to migrate naturally, whereas those at the equatorial periphery may be selected, because they are the most at risk from CC (Hampe and Petit, 2005). Understanding the adaptive significance of intraspecific genetic structure may be impossible without large-scale experiments (Davis and Shaw, 2001), although continued developments in phylogeographic research is gradually making the process of selecting suitable source populations less problematic (McLachlan *et al*, 2007).

Hunter (2007) categorises MR candidate species as those that have a high probability of extinction due to CC, limited vagility, and those that play important ecological roles. The complexity of identifying the relative impacts of various factors threatening a species makes it difficult to be entirely confident that CC is the specific reason causing a species to decline. MR should not be seen as the easy option, relocation should not be undertaken to avoid addressing the threats that pose risks to species within their existing location (Hunter, 2007).

Species that are unable to disperse due to limited vagility or those that have no opportunity to migrate because of habitat fragmentation, as well as being at risk from CC, are considered to be prime candidates for MR. As previously discussed, species that play major ecological roles may help restore ecosystem function and increase resilience, but prove a greater risk to move, as they could also have profound reactions to recipient ecosystems (Soule *et al*, 2003). Ecological role is a complex foundation on which to base species selection; abundance, and therefore the strength a species ecological role, can fluctuate over space and time (Jacobson and Dieffenbacherkrall, 1995). Deciding upon suitable MR candidates will undoubtedly present problems, and it is imperative that only taxa that explicitly require MR are selected, to ensure funds are not wasted on undertaking MR, where there are other, more appropriate alternatives.

14. Is Managed Relocation Affordable?

Few reliable figures exist relating to the cost of maintaining biodiversity (Pimentel et al, 2007). Estimates have ranged from $680 million to $42 billion (UNEP, 1992), and cluster around $20 billion (IUCN/UNEP/WWF, 1991; WCMC, 1992; WRI/IUCN/UNDP, 1992). Certain conservation management strategies, such as species relocations, may be perceived as expensive operations (e.g. Kleiman *et al*, 1991; Lindberg, 1992; Rahbek, 1993; Fischer and Lindenmayer, 2000), yet government expenditure on subsidies associated with environmentally harmful practices is far greater than the financial investment required to adequately conserve global biodiversity. Reducing ‘perverse’ subsidy payments by just 10% would provide a substantial contribution to global conservation initiatives (James *et al*, 2001).

James *et al* (2001) calculated the approximate financial outlay required to implement various hypothetical conservation programmes over a period of 30 years. They concluded that conducting a global biodiversity survey would total $4.4 billion (an annual cost of $271 million, including a 5% interest return), extending global reserve networks would cost between $3.4 and $10.7 billion, and the financial investment required to cover additional management costs would be between $1.1 and $3.3 billion. These calculations at first glance appear high, yet when compared to the estimated $240 billion spent annually on global agricultural remediation, the costs for these proposed conservation initiatives are relatively low. The total expenditure on perverse environmental subsidies are estimated between $950 billion (van Beer and de Moor, 1999) and $1,450 billion (Myers, 1998) per year worldwide.

The United Nations Agenda 21 (1993) concluded that an estimated $49 billion is the annual expenditure required to effectively conserve all of the planets forest, marine and freshwater ecosystems; a relatively small outlay in comparison.

Despite this study finding no correlation between mammal relocation costs and GDP, a much broader study by Balmford *et al* (2003) found that the cost benefit ratio (i.e. the financial investment needed to initiate effective conservation programmes in relation to the overall benefit to global biodiversity) was higher in less developed regions with low GNP (Gross National Product), they go on to demonstrate that investment in conservation initiatives is lowest within these countries. Areas shown to have high levels of biodiversity, endemism and threat are generally located in the less developed regions of the world. This highlights that there are potentially beneficial and cost effective conservation programmes that could be initiated (James *et al*, 2001).

The challenge lies with conservation practitioners to redirect government expenditure into sustainable environmental programmes, and away from perverse subsidies (James *et al*, 2001). Despite previous calls for better financial accountability (e.g. Fischer and Lindenmayer, 2000; James *et al*, 2001; Balmford *et al*, 2003) ill-defined costs of conservation management continues to undermine appeals for global conservation programmes. This research has highlighted the problems associated with trying to ascertain individual project costs, and the lack of adequate accounting within many conservation organisations. Better information on the financial investment required to achieve conservation objectives may help stimulate economic support for conservation initiatives, especially if the benefits are considerable and costs are lower than policymakers presume (James *et al*, 2001). Conservationists must lobby governments to achieve the shift in government expenditure that would procure the investment required to adequately conserve biodiversity. An increase in financial backing, could provide funds needed to accurately assess the potential of emerging management practices, helping mitigate against global environmental change and reduce the rates of biodiversity loss.

15. <u>Concluding Remarks</u>

The Canadian Wildlife Service RENEW report (2006) provides a good example of how conservation spending should be recorded. The report reveals that there has been a substantial increase in financial support for species recovery programmes, from just over $10 million in 1999/2000, to over $40 million in 2005/06, and that 83% of funding came from government sources. This could indicate that governments have increased conservation expenditure, possibly in response to international treaties such as the Convention on Biological Diversity (CBD). Improved financial accountability could help galvanize support, strengthen financial backing, and increase the effectiveness of global conservation programmes (James *et al*, 2001). This research shows that relocations can be expensive management strategies; yet the movement of animals between localities continues to be a common practice that organisations are prepared to fund. As knowledge relating to the risks and benefits of MR improves, and if outcomes prove positive, organisations may deem MR to be a plausible conservation strategy for some organisms, and be prepared to invest in its implementation.

The RENEW report (2006) shows that over 50% of species recovery funds are directed at mammals and birds, whereas amphibians, reptiles, plants, and other taxa (with the exception of marine fish) receive disproportionate funding. In terms of financial and logistical viability, mammal species may not be the most suited to MR. The experimental MR of two Lepidoptera by Willis *et al* (2009) established two self-sustaining populations for less than $8,000. Furthermore, the ecology of amphibians, reptiles and plants is better suited to short-term ex-situ conservation, which could help maintain source populations, while potential management options are considered. MR programmes for these organisms may face a variety of potential problems; Invertebrates, and other taxa with small body masses, may be cheaper to relocate, but they are also more vulnerable to predation. Large numbers of individuals may have to be released to counteract predation pressure, which could pose a risk to the recipient region; or an expensive predator eradication programme may be required.

Of the one hundred and thirty threatened mammal species identified by the Alliance for Zero Extinction, the population status of ninety nine (76%) is unknown, and ninety eight (75%) inhabit isolated ecosystems. Conducting population assessments would be time consuming and costly, and those inhabiting closed systems are unlikely to be deemed suitable candidates for MR. The IUCN (2010) calculate that approximately 27% of extant mammals are under some degree of threat, and it is doubtful that 'zero extinction' will be attainable; CC is likely to seal the fate of some species, despite the best efforts of conservationists (Parmesan, 1996; McLaughlin *et al*, 2002; Thomas *et al*, 2004; Moritz *et al*, 2008; Ohlemuller, 2008) . Conventional practices will undoubtedly remain integral to conservation science and management approaches, however innovative strategies must still be developed.

The Millennium Ecosystem Assessment (MEA) (2005) recognised that the services ecosystems provide are integral to human survival, yet fifteen of the twenty four services examined are in decline. Sandler (2010) advocates that MR does not preserve the value of species, and on that basis, should only play a minor part in conservation. However, if MR increases ecosystem resilience, and helps maintain processes integral to ecosystem services, particular species may acquire unforeseen ecological and/or instrumental value within newly emerging systems. The responsibility of demonstrating that the risks associated with MR are relatively low, and the value of candidate species are such that those risks are justified, rests with those advocating MR as a viable strategy (Sandler, 2010).

Deciding whether or not MR should be conducted is evidently a complex problem. Research directed specifically at indentifying the risks, opportunities and alternative approaches, will aid the development of effective MR policies (Hunter, 2007; McLachlan *et al*, 2007). Information obtained from previous research and environmental modelling should be used for general guidance; not regarded as definitive predictions. Exploring various scenarios, implementing experimental management, and monitoring the outcome of specific interventions, will provide an insight into future management methodologies, and enhance the effectiveness of conservation efforts (Cole *et al*, 2008).

MR is a solution that some find daunting, due to its unprecedented approach; but rejecting novel conservation strategies, on the grounds that the associated risks and uncertainties are considered irreducible, hinders the capabilities of conservation practitioners dealing with unpredictable future threats (Fox, 2007; Schwartz *et al*, 2009). In some instances MR will be judged too expensive or precarious; but the application of sound scientific methods will help reduce the risks and financial expense of conducting MR; enhancing its potential as an effective conservation practice (Hunter, 2007; McLachlan *et al*, 2007). Further evaluations of MR strategies will ensure that conservationists are positioned to make informed decisions, before climate driven species extinctions become increasingly prevalent. Prevaricating over a solution, and waiting for the effects of CC to develop, will result in disastrous consequences for the planets biodiversity.

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