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RESTORATION OF THREATENED SPECIES POPULATIONS:

TUATARA REHABILITATIONS AND RE-INTRODUCTIONS

A thesis submitted for the degree of
Doctor of Philosophy in Wildlife Management and Restoration Ecology
University of Auckland
NEW ZEALAND

GRAHAM USHER
October 1999
IN SEARCH OF THE TRUTH

SUCCESS !?
ABSTRACT

The role of scientific theory in encouraging greater efficiency and accountability for the restoration of threatened species populations is assessed for an endemic New Zealand reptile, the tuatara Sphenodon spp. The value of examining assumptions underlying concepts such as 'habitat requirements' theory and incorporating scientific principles into species recovery are discussed and perceived habitat needs for tuatara tested through experimental application.

Species restoration in New Zealand, especially re-introductions, are typically undertaken as one-off, non-replicated, management exercises (trials). The lack of comparative controls for trials means that the reason for success or failure of management actions cannot be identified accurately and therefore, cannot be used to improve the probability of success for subsequent re-introductions. Trials also reinforce conservative re-introductions of species to habitats in which species are known to survive, because the risk of failure is inherently lower than re-introductions to dissimilar habitats where habitat suitability is unknown.

An alternative approach is to plan management actions as experiments. Testing the full range of perceived habitat needs of species as experimental comparisons identifies the relative importance of tested factors (e.g. predators, refuges) for species recovery and identifies new management strategies (e.g. reduced level of predator control). Constructing testable hypotheses for the environmental factors thought to affect the success of restoration projects for tuatara identifies (among others) three factors:

1. absence of the introduced rat, the kiore (Rattus exulans)
2. presence of seabird colonies
3. presence of open canopy forest

The threat posed by kiore to established tuatara populations was investigated by determining the existence and degree of food competition before and after an eradication program for kiore on offshore islands. Kiore successfully out-competed tuatara for favoured food items, but the degree of competition differed between forest types. Competition for food was greater in the early regenerating forest than the mature forest. These data support models which propose that kiore are but one of a number of historical and current environmental factors influencing the persistence of native fauna and
that management tools other than eradication may enable restoration of fauna in the presence of kiore.

To test the importance of two environmental factors, forest development and the availability of refuges, in determining the establishment of new populations of tuatara, a planned experimental re-introduction was conducted on Moutohora Island. Tuatara were released into sites where the forest was young with a closed canopy and older with a more open canopy, and in sites where refuges (seabird burrows) were distributed evenly at high densities and where burrows clumped with few burrows between patches. Vegetation age and burrow dispersion had no measurable effect on the survivorship or condition of tuatara. Although tuatara released in areas where burrows were clumped dispersed significantly further from their release points and continued to disperse away from release sites throughout the study, tuatara in all release sites were considered to still be reproductively viable 16 months after the re-introduction. Therefore, sites which support open and closed canopy forest, and seabird burrows at high and low densities should be considered as habitat options for future re-introductions of tuatara.

Testing habitat needs as well-planned experiments offer more reliable information to guide future re-introductions than that generated by trial releases or releases to similar locations. Future re-introductions of tuatara and other wildlife should be designed as experiments to test declared mixes of habitat factors. This will accelerate species recovery by identifying important habitat prerequisites for re-introduction and management options for achieving these, thus refining criteria used for selecting new sites and increasing confidence, efficiency and accountability of subsequent management actions. An example of designing re-introductions as experiments is included as a plan for the re-introduction of tuatara to Tiritiri Matangi Island.
The last five years and the two years before that spent researching aspects of threatened species management have without doubt been some of the best years of my life. To work on remote New Zealand offshore islands has been fantastic and I have valued the opportunity to involve as many different people in my work as possible. So many people have contributed to this thesis - and I'm going to attempt to name them all. My apologies to those of you whom I missed out.

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DISCLAIMER – The views expressed in this thesis are those of the author and do not necessarily reflect those of the individual contributors.

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OVERALL

Most importantly, thanks to Ruby Jones and John Craig for their encouragement when times were bleak and for their (futile) efforts at keeping me to my ever-increasing deadlines for completing chapters.

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The commitment shown by conservation practitioners to restoring threatened species is nothing short of astounding. Unfortunately, the concomitant degree of biodiversity loss is also astounding. The Draft Biodiversity Strategy for New Zealand (the Department of Conservation (DoC) & Ministry for the Environment (MfE) 1998) merely confirms widely held beliefs that despite decades of conservation action and specifically targeted funding, biodiversity loss continues at an ever-greater rate. Also apparent is the inability of conservation agencies to meet even the most modest of conservation objectives solely through government-sponsored funding. To achieve more meaningful conservation gains will require a smarter approach to managing biodiversity, not necessarily a more intensive or extensive approach.

Prerequisites for smarter conservation are increased efficiency and accountability of management actions. Efficient management results from testing alternative management approaches against each other and produces solutions which are cost-effective and which maximise conservation gains. Accountable management relies upon correctly identifying all relevant stakeholders and incorporating their needs into decisions when determining alternative management strategies. Moreover, accountability requires that decision-making frameworks and subsequent actions correspond to the quality of information available so that decisions made are morally, scientifically and socially defensible.

The application of sound scientific principles to management actions is a key component for assessing the success of conservation outcomes. CHAPTER ONE explores how the use of experimental procedures in threatened species recovery can identify alternative management options and improve the efficiency and accountability of management frameworks and decisions.

Where information of species decline is unreliable and incomplete, yet alternative options for achieving conservation gains have not been explored or assessed, conflict between conservation stakeholders over the most effective approach can occur. CHAPTER TWO provides the first experimental test of dietary interactions between the endemic tuatara Sphenodon punctatus and the introduced kiore Rattus exulans. Impact by kiore on tuatara has long been assumed to occur but has never been experimentally tested. The adoption of widespread eradication programmes for kiore on islands under these assumed interactions has recently lead to conflict between New Zealand's...
principle conservation management authority and others who propose alternative approaches for managing kiore populations whilst also achieving species recovery.

Achieving the most efficient approach to species recovery means measuring comparative conservation gain from viable management alternatives. Underlying assumptions made by managers for habitat suitability of species reintroductions have the potential to unjustifiably reduce the number and range of locations in which effective recovery can be achieved. For tuatara, two environmental factors thought to be important criteria for selecting new re-introduction sites are experimentally assessed in CHAPTER THREE. The importance of adopting experimental approaches to species management is also discussed, as is the importance of directly involving people in conservation to foster awareness and support for conservation initiatives.

Major findings from this research and their application to conservation theory and management are presented in CHAPTER FOUR.

In recognition of the important role that both science and the general public play in ensuring that conservation actions are effective, efficient and accountable, CHAPTER FIVE outlines a working plan for the re-introduction of tuatara to Tiritiri Matangi Island. Tiritiri is an open public access wildlife sanctuary and has the habitat heterogeneity amenable to further exploring the habitat needs of tuatara through well-planned experimental releases.

Throughout this study, data have been gathered on varied topics peripheral to the core experimental sections of this thesis. CHAPTER SIX details three papers or articles prepared as summaries of some of these additional data. A brief list of papers proposed, or under way, for the remainder of this data are also included.

**Thesis Structure**

The thesis is written as a series of stand-alone proto-papers to facilitate publication. Some overlap occurs between chapters, especially in descriptions of tuatara biology and references. At present, Chapter 3 has been submitted to a journal and Chapter 5 to the New Zealand Department of Conservation. Other chapters will shortly be submitted to journals.

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CHAPTER ONE

HYPOTHESIS TESTING FOR
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RE-INTRODUCTION

Abstract
The use of hypothesis testing in species recovery planning is critically examined. Assumptions underlying the concept of 'habitat requirements' of species are discussed and the merits of two different frameworks in which to conduct species restoration - trials and experiments - are compared.

Rehabilitation and re-introduction biology is currently characterised by a general lack of theory and a prevalence of assumptions underlying management decisions. Undertaking management actions as single, one-off, non-replicated events (trials) is the most common approach to effecting species restoration. However, trials discourage investigation of alternatives to currently accepted dogma, narrow management perspectives and provide a misplaced sense of confidence in restoration outcomes. In contrast, well-planned experiments offer increased efficiency and effectiveness for management actions through assessment of species habitat plasticity and the potential for diversification of conservation philosophies underlying re-introductions. Identifying testable hypotheses for the management of tuatara (Sphenodon spp.), an endemic New Zealand reptile, will enable assessment of the merits of re-introductions to public access areas and the identification of environmental factors important for determining population establishment.

Keywords: habitat requirements, rehabilitation, re-introduction, tuatara, Sphenodon, trials experiments, species recovery planning
INTRODUCTION

Conservation biology has long been termed a 'crisis discipline' in recognition of the often pragmatic and intuitive-based management decisions made to remedy species decline. Caughley (1994) describes two disparate threads of conservation biology, one of which considers declining populations and their causative threats. This 'declining population' paradigm is characterised by its strength in conservation practice but lack of underlying theory (Caughley 1994, Sarrazin & Barbault 1996), a characteristic attributed to its origins in wildlife management rather than ecological science (Soderquist 1994). As a consequence, determining cause and effect for declining populations has more often been predicted by association rather than identified through rigorous experimental testing (Caughley & Gunn 1996, Veltman 1996). In the absence of clearly defined hypotheses and tests of these it is difficult to distinguish between multiple factors which may have instigated or exacerbated decline (e.g. Short et al. 1992, Backhouse et al. 1994, Veltman 1996).

In threatened species management, observations of species autecology are used to construct a speculative model. Hypotheses from this model are sequentially generated and tested under controlled experimental conditions to determine the agent(s) of decline. Additional testing can reveal the dynamics of impact (e.g. temporal or spatial patterns) and the level of management intervention required to achieve species recovery. However, restoration programmes for threatened species are, by their very nature, often severely limited by the number of individuals or by the amount of remnant habitat available for controlled manipulations. The consequences of this are two-fold.

First, for populations managed in remnant areas (re-habilitation), the necessity or efficiency of management practices compared to alternative treatments may not be measured because insufficient numbers of animals are available as experimental controls. Small sample sizes and the absence of controls mean that data generated are sensitive to methods of analysis and assumptions underlying the generation of hypotheses. Studies of a declining population of African wild dogs Lycaon pictus suggested causes of local extinction as competition from predators or outbreaks of various epizootics (Ginsberg et al. 1995 and refs therein). However, while these factors were still considered to have contributed to a small and declining population, a re-analysis of the data led Ginsberg and colleagues (1995) to conclude that demographic population factors alone may have been the proximal cause of extinction. In that case, reassessment of the underlying assumptions of the role of ecological factors in the extinction of the population resulted in different conclusions.

That study also noted that identifying causes of decline for small and declining populations in the absence of controls will always be difficult and often impossible. Where contemporaneous controls (comparison of manipulation and control sites at the same time) are not available, retrospective
controls are recommended by some authors (e.g. Armstrong et al. 1994). However, the limitations of from contemporaneous studies (lack of simultaneous controls, replication) and generality of the results to con-specific situations must be acknowledged by adopting cautious management. The validity of single, poorly replicated tests of habitat suitability can be clarified by repeating studies to verify tentative conclusions.

The second consequence of limited numbers of individuals or habitat is that because individuals are few and valuable, conservative management is often seen as the most responsible approach to ensuring quick and effective recovery. Restricting management practices to those which have worked previously, or selecting new sites for species recovery (re-introduction) whose habitat characteristics mirror those of remnant sites, intuitively present the least risk to species and offer the greatest chance of success for managers. However, both conservative management and a reluctance to reliably assess management actions may restrict species recovery by limiting options available to managers.

This paper discusses the assumptions underlying management actions for the rehabilitation of remnant populations and the re-introduction of species to new locations. It explores the merits of incorporating scientific principles of project design into management practices, in particular the use of experimentally-based approaches for species re-introductions, and the implications of hypothesis testing in species recovery for current conservation management. This chapter draws on studies of animals; specific reference is not made to the re-introduction of plants although many of the concepts presented in this chapter (and the following thesis) are likely to apply to plant conservation.

HABITAT REQUIREMENTS THEORY

Determining habitat use and 'requirements' of species is an integral part of threatened species management. Studies of species ecology, behaviour and physiology impart valuable information which guides population management (harvesting, re-introductions), management intervention (such as supplementary feeding or artificial nest boxes) and habitat restoration. Biodiversity management seeks to minimise risk to populations and so predictions of habitat suitability will dictate the type and degree of management intervention undertaken. However, it is important to acknowledge that associations noted between habitat and species, whether based on sound scientific study or on casual relationships, form an underlying model for which assumptions are made and testable hypotheses may be generated (Gray & Craig 1991, Armstrong & McLean 1995). Gray and Craig (1991) discuss three major theoretical assumptions underlying the concept of 'habitat requirements':
1. Optimality

Assumes that the current habitat and behavioural preferences of the species are optimal

2. Genetic determinism

Assumes that the species' behaviour is fixed

3. Historical factors

Assumes that current ecological factors are more important than historical factors

**Optimality**

Habitats assumed to be optimal for yielding, for example, maximum population growth rates may instead represent but one of many similar quality habitats or may in fact be sub-optimal compared to alternative habitats. If resident habitat is considered optimal without testing productivity in other habitats then management actions may hinder, rather than accelerate, species recovery. This is especially the case if 'optimal' habitats are actively managed to ensure persistence of habitat features identified as critical for maintaining current productivity. Saddleback *Philesturnus* spp. were once a common forest bird on the New Zealand mainland, but became extinct in the presence of extensive forest clearance and introduced mammals (King 1984). The North Island species *Philesturnus carunculatus* was restricted to a single offshore island supporting tall mid and late successional forest which was thought to be necessary for successful population establishment (Bell 1989). North Island saddleback were subsequently re-introduced to islands supporting early regenerating and mature forest. Clutch size, clutch frequency and population density of individuals was greater in early regenerating or recently planted forest than mature forest (Craig 1994). Thus, regenerating forest offers equal or greater potential for the establishment of viable saddleback populations and re-introduced populations can quickly act as sources for subsequent re-introductions.

**Genetic determinism**

Determining behavioural plasticity of species can also greatly increase the range of options for conservation managers. Rare species are often restricted to a subset of habitats which are currently or may have historically been available. Inferences made about dietary preferences may not hold true once individuals are introduced to new habitats and food supplies. The critically endangered Mauritius Kestrel *Falco punctatus* was originally thought to be adapted specifically to hunt in closed canopy forest (Temple 1987). Some captive-bred kestrels were released outside their remnant native forest range and into fragmented agricultural habitats comprising primarily exotic trees. Those individuals exhibited a range of behaviours rarely, or never before, seen, including the use of artificial nest boxes, a new range of hunting behaviours, and the exploitation of new exotic food sources (Cade & Jones 1993).
Imprinting can also be used to increase the chances of survival for species. Efforts to assess the potential of predator-inhabited islands for saddleback re-introductions have utilised the social transmission of roost site preferences passed from adult saddleback to their young (Lovegrove 1992). Predator-proof artificial roost boxes were placed in source habitat prior to translocation. Consistent users of roost boxes were then translocated, along with control non-box-users to a rat *Rattus norvegicus*-inhabited island. Levels of depredation by rats on roost-box users were considerably less than on non-box users, but population modelling still indicated gradual, long-term, population decline (Lovegrove 1992).

Behavioural plasticity and transmission of behaviours can be equally important tools for planning management in remnant habitat. Remnant sites are often least disturbed habitats which generally undergo relatively few structural changes over time but conservation sites in transitional habitats can expect considerable changes to habitat characteristics. Knowledge of the ability of species to survive and reproduce in alternative environments may allow modification or even discontinuation of current management practices (e.g. the level of predator control required) and also help identify new habitat for addition to existing conservation areas. Under ideal conditions of low population density and free choice of habitat type, it is intuitive that species would choose habitats which are most preferred and thus generate maximum population growth. However, preferred habitats may not necessarily be related to optimality but may, in fact, reflect the behavioural conditioning of founders (Gray & Craig 1991). Therefore habitat used may be sub-optimal but preferred, rather than that required to maximise reproductive output.

**Historic factors**

Consideration of historic ecological patterns can greatly improve the potential options for species management. Contemporary patterns of habitat use by species may not represent the full range of habitats in which they were once distributed, nor the range of habitats in which they can do best. Comparison of historical information (anecdotal, archaeological) with present range can identify new sites in which remnant populations may still exist, or identify sites in which translocated individuals can be managed more efficiently with comparative conservation gains. The endangered New Zealand wattled crow, the Kokako *Callaeas cinerea wilsoni*, is restricted to tall, native forest on mostly mainland locations (Rasch 1991). Correlations between kokako feeding behaviour, diet and reproduction with physical characteristics and productivity of fruit-bearing tall forest trees (Rasch 1991 and refs therein) suggested tall forest as a critical habitat criterion for future reintroduction sites (however, see Craig (1991) for contrary view). Recently however, kokako superfluous to a captive-breeding programme were released onto an island supporting mainly naturally regenerating and
young planted forest but with some tall forest remnants. The translocated birds used both remnant mature forest and regenerating habitat and the single breeding pair of birds on the island established a territory in regenerating forest and successfully raised two chicks to independence (Ruby Jones pers. comm.).

Historic considerations are also valuable when determining the significance of current species interactions. Assessments of predator or competitor impacts on species often assume that the degree of impact is the same now as it was in historic times or will remain the same in the future without management intervention. Therefore, under these assumptions, either predator eradication or sustained control is necessary to achieve and maintain recovery gains. In some cases recovery may be possible in the presence of predators if predator densities have declined since historical times (Veitch 1994) or habitat factors contributing to predator impact have now changed. Well-planned tests of these hypotheses backed by an in-depth understanding of impact dynamics of predators or competitors may reveal cost-effective solutions for recovery which may mean that expensive pest control or eradication programmes are not necessary.

Ethical debates over the most appropriate method of pest control are often raised by non-governmental conservation groups, but are rarely considered by government conservation agencies. Increasingly effective eradication programmes, especially on islands, can now be undertaken with a high degree of success and security from re-invasion. However, reliance on a single management solution, such as eradication, can limit options when perceptions of the value of pest and native species differs between conservation stakeholders. It also discourages consideration of alternative pest control strategies, which may result in a similar conservation gain to threatened species. Management of the Polynesian rat or kiore *Rattus exulans* on islands inhabited by the rare endemic reptile, the tuatara *Sphenodon* spp., illustrates these problems well.

**Impacts of kiore on tuatara**

Debate over the best course of action to protect wildlife, but also to protect kiore, has been exacerbated by a lack of knowledge about how historical factors, combined with the presence of kiore, resulted in the probable extinction or decline of some wildlife. Strong circumstantial evidence links the kiore to the decline or local extinction of invertebrates (Ramsay 1978), lizards (Whitaker 1978, Towns 1991), birds (Atkinson 1978) and tuatara (Crook 1973, Newman 1988, Cree & Butler 1993, Cree et al. 1995). However, little is known of the mechanism by which kiore threaten native wildlife and, indeed, whether kiore alone are the only or most important factor (Craig 1986). Kiore are a culturally important species to some Maori (e.g. Ngatiwai 1995) and Maori are important stakeholders in New Zealand conservation. Islands inhabited by kiore as the only introduced
mammal also represent a stage of New Zealand’s ecological history that reflects New Zealand when Europeans arrived. Preservation of this ecological history is seen as important by some scientists (J. Craig pers. comm.).

Three different models for the impact of kiore on wildlife have been proposed (Table 1). The Predator Model assumes that kiore is the most important factor, irrespective of habitat or population characteristics. Using this model, eradication is the only effective tool for restoring wildlife populations. However, the drawback of relying on only one management solution (eradication) has become apparent as Maori (e.g. Ngatiwai 1995) and some conservation biologists (e.g. Craig 1986, Veltman 1996) have raised questions about the quality of information which drives current eradication programmes and links kiore alone to fauna decline. This also makes it difficult to assess models which propose solutions other than the current eradication-based management strategies for wildlife protection and enhancement (Table 1).

Table 1. Assumptions of models proposed for the impact of kiore on tuatara and other wildlife.

<table>
<thead>
<tr>
<th>Model</th>
<th>Core principle</th>
<th>Assumptions</th>
</tr>
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<tbody>
<tr>
<td>Predator Model</td>
<td>past, current &amp; future impacts threaten wildlife persistence.</td>
<td>Effect of kiore greater than all other factors - food availability, habitat change, population variability</td>
</tr>
<tr>
<td>Harvest Model</td>
<td>impact is only recent - impact historically regulated by human harvesting.</td>
<td>Intensity of impact is positively related to density of kiore</td>
</tr>
<tr>
<td>Historic Impact Model</td>
<td>impact is most intense in the past - degree of impact reduces as forest regenerates.</td>
<td>Intensity of impact is positively related to density of kiore Select resident or re-introduced wildlife can persist if above threshold densities in presence of kiore</td>
</tr>
</tbody>
</table>

In the Harvest Model, Maori contend that species decline or extinction today is a recent phenomenon produced by changes in human management of kiore populations on islands (Hori Parata pers. comm.). Kiore were an important food and trade item for Maori (Haami 1992, Roberts 1995) and were harvested on a regular annual basis from islands and mainland sites (Hori Parata pers. comm.). That model proposes that a cessation of harvests some 150 years ago, associated with the arrival of European settlers, resulted in densities of kiore unregulated by human harvesting on islands and greatly increased levels of predation on native wildlife. Re-instatement of a long-term,
regular harvesting regime is predicted to suppress kiore numbers so that native fauna could be re-introduced to islands and remnant populations recover.

The Historic Impact Model proposed by Craig (1986) is an alternative or additional model to the Harvest Model and is based on studies of kiore population dynamics on islands in different habitats. Kiore population densities cycle with a regular annual pattern (Bunn & Craig 1989, Moller & Craig 1987, Roberts & Craig 1990) where the availability of high protein seeds and fruits during spring and summer regulate the amplitude and duration of population cycles (Craig & Bunn 1989). All New Zealand islands are considered to have suffered human-induced habitat modification (Hayward 1986) since Maori (approx. 1000 years ago) and European (150 years ago) settled New Zealand. Changes in the seasonal availability of seeds and fruits as vegetation regenerates (Speed 1986) and matures successively reduces peak kiore population densities, dampens population fluctuations (Craig 1986) and reduces individual litter size (Roberts & Craig 1990). Kiore will preferentially forage on fruits and seeds (Roberts & Craig 1990, Ussher submitted) but switch increasingly to less favoured vegetation and animal foods as favoured foods become scarce. Craig (1986) proposes that, because of seasonal differences in the availability of favoured foods during the year and between habitats and the subsequent effects of this on kiore feeding and population dynamics, kiore impact on native fauna will be more severe in regenerating habitats than mature habitats.

Both of these latter models imply that past levels of impact by kiore may not accurately reflect present levels. Moreover, the Historic Impact Model also implies that restoration of some components of native fauna may be possible in the presence of existing kiore population levels in mature habitats. Studies of tuatara dietary interactions with kiore provide quantitative support for hypotheses generated by the Historic Impact model (see Chapter 2).

In summary, evaluating potential underlying assumptions can open new directions for species management. New management actions for countering species decline will only separate underlying causes if undertaken as well designed experiments. Without this approach, the cost-effectiveness of the actions and their probability of success for future applications cannot be known with confidence. While rehabilitation of populations would benefit from greater experimental and theoretical considerations of proposed management actions, it is the approaches taken to the restoration of species beyond their current ranges and habitat usage, that can highlight the application of these principles to species recovery.
RE-INTRODUCTION BIOLOGY - A TOOL FOR A CRISIS DISCIPLINE

History and trends in re-introductions in New Zealand

Threats facing remnant populations or inadequate area to ensure long-term conservation of remnant populations has seen the development of species transferral techniques from one location (source) to another (destination). Terminology for species transferrals used in this thesis is defined below and follows that used in the general published literature. This differs somewhat from the terminology used by The World Conservation Union (IUCN) (1998).

1. Transfer/ Translocation deliberate movement of a species from one location to another
2. Reintroduction an attempt to establish a species in an area which was once part of its historical range
3. Introduction an attempt to establish a species outside its recorded distribution
4. Augmentation addition of individuals to an existing population of con-specifics

Some of the first recorded species transfers in New Zealand were in the 1890s by Richard Henry to save mainland Kakapo Strigops habroptilus and Kiwi Apteryx spp. by establishing populations on islands in the Dusky Sound (Atkinson 1990). Overall, Richard Henry undertook more than 700 native species transfers between 1894 and 1908 (although this includes multiple transfers to a site, Hill & Hill 1987).

More recently, both in New Zealand and internationally, the use of re-introductions has steadily increased, although not at levels recorded in Richard Henry’s days. Between 1973 and 1981 the average number of translocations (then defined generally as the movement of species from one location to another) per agency doubled for North America, Australasia and Hawaii (Griffith et al. 1989). In New Zealand the number of fauna transfers has fluctuated over the last 30 years (Fig. 1), reaching a peak of 47 transfers of species to separate locations between 1960 and 1964 and decreasing to a low of 15 between 1975-1979. The popularity of transfers as a conservation measure is again increasing, with the last 5 years seeing the most transfers undertaken in recent history, i.e. since 1964. Traditionally dominated by avifauna, transfers in the last 15 years have started to include invertebrates, amphibians and, especially, reptiles (Fig. 1). Indeed, reptile re-introductions have increased 6-fold over the last 5 years compared to the previous 10 years, and transfers during 1995 - early 1999 comprised over one third of all re-introductions for that period.

Planning restoration projects on wider ecological principles will see better co-ordination between practitioners in reptile, invertebrate and avifauna fields and a greater focus on re-introduction time-
Figure 1. Relative composition and numbers of different species within major taxonomic groupings transferred to one or more locations (re-introduction, conservation introduction or supplementation) in New Zealand between 1960 and 1999.
frames for species at different trophic levels within communities. There are already indications of this in recent trends in New Zealand re-introductions. The average number of locations to which a species is transferred has decreased significantly over the last 10 years compared to most previous time periods (Table 2). Additionally, the number of separate transfers of a species to a given location has also decreased (relative size of standard errors in Table 2). Current trends for invertebrate, amphibian and reptile transfers also reflect these patterns, with species being transferred to few locations and usually as one-off exercises (Table 2). The diversity of species transferred has increased nearly 5-fold since 1960 (Table 2) and the significant trend towards greater diversity as time progresses (Spearman’s $r^2 = 0.89$ n=8 p<0.005) suggests that species transfers will continue to be both more frequent and include a greater diversity of species.

The decrease in the rate of new transfers for individual species and corresponding increase in the diversity of fauna being transferred can be attributed to a combination of more stringent approval procedures for re-introductions, more targeted ecosystem-based restoration re-introductions (e.g. Department of Conservation (DoC) 1999) and greater predator control technologies enabling restoration of more vulnerable species. Changes in restoration philosophy from single to multiple species-driven restoration projects (e.g. Towns et al. 1990) will see increased diversity amongst re-introductions continue.

The dominance of a few bird species among transfers is clearly shown by the high variability in number of transfers per species to new locations (Table 2) between 1960 and 1989 compared to the last 10 years. Indeed, since 1960, 70% of all bird transfers were undertaken for only 9 out of 43 bird species. This trend can be explained by the variable vulnerability of different species to predators. In the 1960s-1970s predator control was limited to large species (e.g. cats) and so birds which could exist in the presence of European rats (*Rattus rattus* and *Rattus norvegicus*) on the mainland (e.g. Brown Teal *Anas aucklandica chlorotis* and North Island Weka *Gallirallus australis greyi*) and kiore on islands (e.g. N. I. saddleback), were transferred frequently and to many locations. Improved rodent control techniques now allow removal of all mammalian predators from islands and effective control of predators in mainland ‘island’ sites, enabling a greater diversity of fauna to be managed. However, re-introductions for many native fauna are still in the assessment stages (especially invertebrates) and more extensive use of re-introduction as a restoration technique will depend on the success of these initial efforts.

Many re-introductions now recognise that conservation goals should encompass not only biological objectives but also socio-political objectives. Nowhere is this demonstrated more than in the re-introduction of the endangered Brothers Island Tuatara *Sphenodon guntheri* to a public access island in the harbour of New Zealand’s capital city. This tuatara is found on only two small islands,
one of which is a recent re-introduction and numbers only 38 individuals (Nelson 1998). However, the Tuatara Recovery Plan (Cree & Butler 1993) recognised the critical role that public access and appreciation of wildlife play in fostering widespread support for long-term conservation of species. The broadening of re-introduction objectives will continue as conservation stakeholders call for re-introductions which are orientated more towards meeting cultural, scientific, historic, educational, aesthetic and public accessibility objectives.

Table 2. Trends in the number of species transferred between 1960 and present (mean number of transfers of specific species to different transfer sites ± standard error) in New Zealand. Transfer defined by receiving site, not by frequency of transfers of founders to site. Number of different species per time period for respective groups given in brackets. For birds, significant (p<0.05) differences in number of transfers per species from the 1990-1999 (mean 1.5 ± 0.2, n= 35) are indicated above respective column (SIG = significant; NSIG = not significant). Data from Girardet & Veitch (unpublished), IUCN Re-introduction Specialist Group Australasian database, Chris Green (pers. comm.) and Towns et al. (in press).

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</tr>
</thead>
<tbody>
<tr>
<td>BIRDS</td>
<td>6.7 ± 4.9</td>
<td>4.3 ± 2.4</td>
<td>3 ± 1.4</td>
<td>1.7 ± 0.7</td>
<td>2.4 ± 0.4</td>
<td>2.5 ± 0.9</td>
<td>1.4 ± 0.2</td>
<td>1.6 ± 0.3</td>
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<td></td>
<td>(7)</td>
<td>(6)</td>
<td>(6)</td>
<td>(9)</td>
<td>(16)</td>
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<td>(18)</td>
<td>(17)</td>
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<tr>
<td>REPTILES</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1 ± 0</td>
<td>1 ± 0</td>
<td>1.4 ± 0.2</td>
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<td>(2)</td>
<td>(3)</td>
<td>(12)</td>
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</tr>
<tr>
<td>AMPHIBIANS</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1</td>
<td>1</td>
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<tr>
<td>INVERTEBRATES</td>
<td>-</td>
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<td>2</td>
<td>1 ± 0</td>
<td>1 ± 0</td>
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<td>(1)</td>
<td>(3)</td>
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<tr>
<td>TOTAL SPECIES</td>
<td>7</td>
<td>6</td>
<td>6</td>
<td>9</td>
<td>16</td>
<td>18</td>
<td>25</td>
<td>32</td>
</tr>
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</table>

Attributing reasons for success or failure of re-introductions

The history and nature of species recovery efforts (the crisis philosophy) means that most re-introductions have been conceived as one-off management exercises (Armstrong & McLean 1995).
Many authors note the lack of post-release monitoring which characterises many re-introductions (e.g. Copley 1994), making it difficult to ascertain the reasons for success or failure. For tuatara, a number of re-introductions were attempted in the early 1900s (e.g. Hislop 1920), but for unknown reasons, populations failed to establish. The absence of detailed records or post-release monitoring for these efforts means that little useful information is available to guide further tuatara re-introductions.

As a consequence of poor information, species re-introductions have, until recently, been characterised by a lack of cohesive principles. Griffith and colleagues (1989) provided the first detailed attempt at determining causes of success or failure through retrospective analysis of re-introductions. That analysis identified general factors associated with success, including numbers and habits of founders and similarity between source and destination habitats. Although the importance of exploring the values of dissimilar habitats as destinations was noted, the broad, prescriptive results of the analysis encourages wildlife managers to engage in conservative approaches to species re-introduction. Indeed, the good rate of success for New Zealand re-introductions compared to those elsewhere has been partially attributed to managers avoiding re-introductions where the outcome is uncertain (Armstrong & McLean 1995). It is not the objective of this review to detail demographic, genetic or behavioural components assumed to be of importance in determining re-introduction success, nor effects of re-introduced species on their host environment. For that information the reader is referred to Armstrong & McLean (1995), Simberloff (1988), Soulé (1987) and Copley (1994) - but see also Lande (1995) for revised estimates of effective population size to maintain genetic integrity of small populations and Craig (1994) for arguments about the appropriateness of estimates to New Zealand conservation.

The trend towards more quality information and greater accountability in re-introductions has generated a substantial literature on guidelines for undertaking re-introduction programmes (e.g. Kleiman et al. 1994, IUCN 1987, 1998). The policy guidelines for species re-introductions prepared by the IUCN’s Re-introduction Specialist Group (IUCN 1998) emphasise the need for reliable information on potential biological threats and assessment of habitat preferences and requirements prior to undertaking a re-introduction. These guidelines can have serious implications for planning of re-introductions. First, they suggest that a pre-requisite for re-introductions is the reliable identification and control of factors threatening remnant populations. Second, they reinforce conservative procedures for selecting sites for species re-introductions and discourage consideration of potential underlying assumptions (discussed earlier). For example, the 1986 re-introduction plan for the Greater Stick-nest Rat *Leporillus conditor* (Copley 1994) set stringent selection criteria, based on habitat associations of remnant populations and requirements for an absence of potential predators, which considerably reduced the number of potentially viable re-introduction sites.
Overall, the implications of these principles exemplify the increasing calls from conservation biologists for a more scientific approach to re-introductions. While there have been frequent calls for greater use of experimental principles in identifying causes of decline (e.g. Caughley 1994; Veltman 1996), there have also been pleas in recent literature for greater inclusion of scientific procedures for the identification and testing of potential re-introduction sites (May 1991, Short et al. 1992, Armstrong et al. 1994, Soderquist 1994, Southgate 1994, Armstrong & McLean 1995). For some authors (e.g. Kleiman et al. 1994) all re-introductions should incorporate a 'research and development' component, while others consider a re-introduction a failure unless useful knowledge is produced from the process to assist with long-term conservation of the species (Southgate 1994). Many authors consider that the future credibility of re-introduction biology as a viable, cost-effective and efficient conservation technique depends upon its maturation as a science. Only by acknowledging the underlying models and assumptions associated with selection of new sites and planning re-introductions as well designed experiments will this metamorphosis from a predominantly ad hoc, crisis discipline be achieved.

INCORPORATING SCIENTIFIC PRINCIPLES INTO RE-INTRODUCTION PRACTICE

Experimental and trial re-introductions

In ecology, experiments are usually defined as the controlled manipulation of environmental factors to investigate hypotheses generated from a predictive (speculative) model. Experiments have also been viewed under different criteria, including deriving conclusions purely from deductions or correlations or gaining support for hypotheses from observations (see Armstrong et al. 1994). These latter definitions are usually considered to be trials rather than true experiments because they do not have comparative controls and the information derived from them is unable to be replicated with a measurable degree of certainty. In re-introduction biology, many projects are considered by their authors to be experimental releases, but are in fact trials. Trials have value in exploring new release procedures or monitoring techniques (Armstrong et al. 1994), but, while also useful for generating new hypotheses, should not be relied upon to provide sound conclusions on which to base future management decisions.

However, trials constitute informal experiments in their own right, and, together with individual experience and deductive reasoning, can provide a powerful tool for advancing species recovery when combined with true experimental re-introductions. The release of a small number of less
Valuable individuals (e.g. excess breeding stock or unpaired/older animals) in a 'trial' re-introduction can provide valuable information on gross habitat suitability and help refine later, more substantial experimental re-introductions (Fig 2). Even if a trial re-introduction is a resounding success, this should only be considered the first stage in determining the suitability of habitat for the species. Because the reason for success or failure cannot be discerned with confidence from trials, outcomes are limited to either omitting that specific habitat from future plans (Fig 2) or including it with no confidence of success in successive re-introductions within that habitat type.

In contrast, experimental re-introductions can identify important variables determining success or failure and measure the likelihood of future successes within habitats supporting that factor. Moreover, the degree of management intervention required to ensure success (in the presence of a given factor) can be determined through successive translocations if initial efforts fail (Fig 2). Once variables determining re-introduction success are known, selection criteria for future re-introductions can be based on components of habitats rather than habitats themselves (as for trials). This is especially pertinent to the changing emphasis in New Zealand re-introductions from mobile bird species to less mobile invertebrates and reptiles. Whereas mobile species may move to more favourable micro-habitats or habitats which differ from their release sites, less mobile species cannot and are dependent on decisions made by managers for site suitability.

Compared to trials, experimental re-introductions enable longer-term gains in knowledge of habitat needs and certainty of success (Fig 3). Because the reason(s) for outcomes from experimental re-introductions are known, successive transfers (1-6 on Fig. 3) to test new habitat variables can build incrementally on past knowledge. This also means that the environmental variables that comprise criteria for re-introductions can be constantly refined and applied independent of habitat type (c.f. for trials habitat types are the unit of selection), greatly expanding the number and variety of viable or potentially viable re-introduction sites (as designated by the expanding cone Fig. 3). Moreover, there will be a correspondingly greater increase in the probability of success for experimental re-introductions to new habitats or those which contain known habitat needs than for trials.

In contrast, trials (Fig. 3; I-VI, A-F & a-f) will always return a lower rate of information gain and reduced certainty of success per re-introduction to the same habitat because factors determining success are confounded with other habitat variables. Successive re-introductions to the same habitat type for trials will produce some gains in information (especially if fates of founders are well-monitored following release), but trial re-introductions to different habitats (as denoted by the individual lines on Fig. 3) cannot build on past experience because the importance of predicted critical environmental variables common across more than one habitat type are unknown in new habitats.
Figure 2. Comparison between the design and management outcomes of experimental versus trial re-introductions of species to new habitats. The solid, bold line represents the level at which preliminary information from trial re-introductions can be further assessed with confidence by experimental re-introductions.
Merits of designing re-introductions as experiments

The benefits of undertaking a re-introduction as an experiment compared to a trial outweigh the costs considerably (Table 3) and are generated throughout the duration of the re-introduction project. Benefits are accrued in the early planning stages because examination of models encourages consideration of underlying assumptions of the species' current distribution (see earlier), scrutiny of alternative management options, and helps identify and prioritise critical management questions (Soderquist 1994, Table 3). Moreover, identifying specific testable hypotheses encourages planners to refine monitoring methodologies so that appropriate data are collected (Soderquist 1994).

Experimental re-introductions often require greater monitoring effort than trials, but offer opportunities to reduce this through inter-agency co-operation. Experiments endeavour to distinguish the outcomes of each individual test sites, but for trials it is the overall outcome from the entire release that is important. Additionally, preparing re-introductions as experiments may require a greater number of re-introduction sites (as controls and replicates) and founders than planned under a trial release. Increased planning and monitoring workloads (and related increases in short-term costs) for projects can be offset by fostering closer ties with internal or outside agencies who can provide personnel, expertise and finance. The regular threatened species re-introductions to Tiritiri Matangi Island are funded largely by sponsors and donations organised by a non-governmental, non-profit support group. Post-release monitoring is largely undertaken by university graduate students whose research is partly financed by the non-profit group, partly by the university and partly by the students themselves.

Table 3. Benefits and costs of planning re-introductions to new habitats as scientific experiments. Benefits are accrued throughout the planning and implementation process, including generating long-term reduction in financial outlay for future re-introductions.

<table>
<thead>
<tr>
<th>BENEFITS</th>
<th>COSTS</th>
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<tbody>
<tr>
<td>Identify critical management questions</td>
<td>Increased planning for re-introductions</td>
</tr>
<tr>
<td>Refine protocols to monitor re-introductions</td>
<td>Increased monitoring effort</td>
</tr>
<tr>
<td>Foster intra and inter agency co-operation</td>
<td>Short-term costs of re-introductions increased</td>
</tr>
<tr>
<td>Produce reliable knowledge of outcomes</td>
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<tr>
<td>Gain knowledge at a greater rate</td>
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<tr>
<td>Ensure efficient adaptive management</td>
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<tr>
<td>Expand management options</td>
<td></td>
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<tr>
<td>Long-term costs of re-introductions reduced</td>
<td></td>
</tr>
<tr>
<td>Develop/ refine ecological theory</td>
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</table>
**Figure 3.** Diagrammatic representation of the relative gains in knowledge and certainty of future success for experimental re-introductions and trial re-introductions.
Experimental approaches to re-introductions offer large gains in the reliability of knowledge compared to information generated from trials. This allows management organisations to adopt the more modern management approach of 'the learning organisation' (Senge 1990) including the idea of 'adaptive management' (Hobbs & Norton in press) and generates a greater rate of knowledge of species' plasticity (Fig 3). As a result, options for managing species and their constraining environmental factors (such as predators or roost availability) are greatly increased. Although project costs for initial investigations of habitat suitability are high for experimental re-introductions (more sites, greater monitoring effort), long-term costs per re-introduction will be lower as the risk of future failure (resource consumption with no net benefit) is reduced and cost-effective management options (e.g. partial, intermittent predator control) for sites can be identified before re-introductions occur. However, to fully capitalise on the potential benefits of the experimental approach, management organisations will need to adopt more open systems for generating and testing hypotheses. The first step in priority setting will be to seek information on potential factors from all stakeholders to ensure that the eventual hypotheses and places for testing are built from all available knowledge rather than just building on existing in-house, ad hoc beliefs. Without achieving this, implementing experimental approaches to restoration will merely test individual managers' beliefs and fail to resolve existing management issues (e.g. the impact of kiore on wildlife, see previous).

Management of takahe Porphyrio mantelli, a large, flightless gallinule of New Zealand, illustrates this well. Studies of morphology and feeding behaviour were interpreted as optimal adaptation to its remnant alpine grassland habitat (Mills et al. 1984). However, sub-fossil remains of takahe are found throughout New Zealand (Atkinson & Millener 1991), a pattern attributed to the greater distribution of alpine habitat during earlier ice ages (Mills et al. 1984). Historic distributions were also interpreted as demonstrating that takahe historically utilised forest and lowland systems (Beauchamp & Worthy 1988) and that their present distribution is a refuge area from hunting and predator effects rather than due to climate change (Beauchamp & Worthy 1988). Because alpine habitat was seen as the only management option, conservation effort there has been intensive over large areas and expensive. Takahe populations in remaining alpine habitat continue to fluctuate around historic levels despite ongoing management of known predators and competitors, extensive egg manipulation, and artificial rearing and release to the wild (Clout & Craig 1994).

Release of takahe into habitats other than alpine grassland and cost analyses in these different habitats indicates that exploring alternative management options can give successful biological and financial outcomes. Takahe have recently been transferred to predator-free islands supporting lowland grassland and forest habitat. Individuals there produce a comparatively greater number of clutches per season, have a much higher rate of yearling survival and a comparative rate of adult
survival than in alpine managed areas (Clout & Craig 1994). Intensive management will be required to maintain or increase populations in alpine areas but little management, other than management as meta-populations (Craig 1994), is needed for island populations. A cost analysis of using island-bred takahe to stock alpine areas rather than captive reared birds indicated an 80% reduction in current management costs ($7,800/ takahe captive reared vs. $1,300/ takahe island-bred; Dawson 1994). Further, if overall conservation of takahe is considered costs of establishing and managing island populations would be far less expensive than producing and managing takahe in their refuge alpine areas. Indeed, a past trial to establish takahe in a new alpine habitat is estimated to have cost $0.5 million with no discernible net conservation benefit (Craig 1997). Moreover, the cost of additional takahe on islands will decrease incrementally but mainland populations will still require the same or greater level of management to maintain or increase population size.

The DoC has recently reaffirmed its commitment to managing takahe in both remnant mainland alpine habitat and on offshore islands (Crouchley 1994). The most effective and efficient approach for meeting management goals will be to compare options for future releases of takahe as well-planned experiments. Questions of the source of founders can be assessed by comparing cost and population persistence using island and captive-reared takahe to stock new or existing mainland sites. Likewise, the comparative merits of creating new populations can be tested by conducting simultaneous releases into alpine and island habitats.

The benefits of designing re-introductions as experiments are not only limited to applied ecology and management. Considerable opportunity exists for co-operation between wildlife managers and theoretical ecologists to advance or refine current ecology theory. Outcomes of well-planned experimental re-introductions can provide valuable information to test models of community processes, population dynamics and genetics and individual behaviour (Sarrazin & Barbault 1996). Indeed, as those authors indicate, because the integration of previous results into models is a critical concept of adaptive management, theoretical and applied ecology complement each other in the context of experimental re-introductions.

**Re-introductions of tuatara**

The management of tuatara demonstrates how priority research areas can be incorporated into a framework of experimental comparisons to refine site selection criteria and promote efficient management. Little is known of the habitat use and needs of tuatara, but a considerable amount of information exists for their current and historic distribution and habitat associations. Management questions range from the basic habitat requirements to ensure re-introduction success to how...
captive rearing procedures will influence the probability of founder survival and population persistence (Table 4).

Control programmes for introduced mammals such as the kiore provide the opportunity to experimentally test the degree of impact on tuatara populations and possible alternatives to eradication (see previous). Assessing the degree of management intervention required to establish new populations of tuatara will be important when new populations are established in mainland locations where pest eradication is not possible and sustained control is the only option. Similarly, all planned or implemented re-introductions (6 islands) offer the opportunity to experimentally test the threat posed by ground-feeding birds (such as Kiwi Apteryx spp, takahe, pukeko Porphyrio porphyrio melanotus, and weka Gallirallus australis subsp.) and morepork Ninox novaeseelandiae novaeseelandiae on recruitment of young tuatara into new populations.

Adopting experimental procedures for re-introductions will also allow tests of the importance of vegetation type and availability of refuges for new populations. Remnant populations of tuatara are generally found in association with high densities of seabird burrows and in mature or advanced regenerating forest. The Tuatara Recovery Plan (Cree & Butler 1993) reflects this by identifying the presence of seabirds and open forest as preferred characteristics for re-introduction sites, although their importance in determining re-introduction success has not been tested.

Table 4. General areas in which hypothesis testing will be useful for the restoration of tuatara populations and the biological significance of these areas for tuatara population establishment. Experimental options are given for testing specific management alternatives.

<table>
<thead>
<tr>
<th>Hypotheses</th>
<th>Biological significance</th>
<th>Tests</th>
</tr>
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</table>
| Introduced mammals threaten population establishment or persistence | -Predation of adults, juveniles and eggs or competition with founders/ persistence of remnant populations | -re-introduction to sites with/without mammals  
1. test presence and mechanisms of impacts  
2. determine level of management required to ensure tuatara persistence |
| Some native birds threaten population establishment | -Predation/ competition & long-term population persistence | -re-introduction in presence & absence of ground-nesting birds (takahe, kiwi, weka, pukeko) & morepork, kingfishers |
| Tuatara threaten persistence of resident native fauna | -Predation on resident fauna  
- Presence of tuatara preclude re-introduction of other fauna | -test effects (positive or negative) of tuatara re-introduction on resident fauna  
-test effects of established tuatara on re-introduced native fauna (e.g. invertebrates) |
| Tuatara can't establish in early forest types or closed canopy forest | -Food availability | -monitor tuatara diet, feeding ecology and body condition of re-introduced tuatara in different forest types, successional stages of forest |
| Released tuatara need abundant natural burrows/refuges to prevent excessive dispersal | -Rate of cutaneous water loss/impact on animal condition | -Thermoregulation for feeding |
| Released tuatara need abundant natural burrows/refuges to prevent excessive dispersal | -Dispersal of founders from release sites/lower long-term population reproductive viability | -monitor tuatara dispersal in sites where refuges are frequent, infrequent, clumped and evenly distributed |
| Public presence threatens or derives no benefit from increased access to translocated species | -Enhanced advocacy, public support | -test use by and dispersal from artificial refuges by founders |
| Wild-caught founders more likely to survive than captive-reared founders | -Anti-predator behaviour/mortality of founders | -determine maximum distances for dispersal before separation significantly reduces reproductive potential of individual tuatara |
| Adult founders more likely to survive than juvenile founders | -Mortality of founders | -re-introduce tuatara into public access and non-access sites to test |
| Gender ratios of clutches will differ markedly with incubation temperature | -Speed of population growth | 1. people impact on tuatara behaviour |
| Gender ratios of clutches will differ markedly with incubation temperature | -Sustainability of source populations | 2. public satisfaction, awareness increased? |
| Gender ratios of clutches will differ markedly with incubation temperature | -Model harvest limits from source islands | 3. does public access = increased founder mortality (e.g. poaching) |
| Gender ratios of clutches will differ markedly with incubation temperature | -Sex of embryo may be temperature dependent | -test survival of na"ive and predator-wise animals |
| Gender ratios of clutches will differ markedly with incubation temperature | -Gender bias for captive breeding of relict tuatara | -test mortality of captive-bred vs. wild caught adults/ juveniles |
| Gender ratios of clutches will differ markedly with incubation temperature | -Speed of population growth | -test mortality of juveniles vs adult founders |
| Gender ratios of clutches will differ markedly with incubation temperature | -Sustainability of source populations | -test survivorship of juveniles in established populations |
| Gender ratios of clutches will differ markedly with incubation temperature | -Model harvest limits from source islands | -model harvest limits from source islands |

Studies of tuatara ecology, behaviour and physiology provide fertile ground for postulating the possible influence of these factors on re-introduction success. Open canopy forest may influence tuatara breeding success and hunting activity and increase the risk of desiccation. Most studies of tuatara reproductive biology have been undertaken on a single island - Stephens Island. There,
female tuatara do not nest in the forest but rather nest in pasture areas where soil temperatures are greater (Thompson et al. 1996). Eggs placed in experimental nests in the forest do not hatch (Cree et al. 1989), and so breeding success may be lower in closed canopy habitats than open canopy habitats where soil temperatures are greater. Tuatara thermoregulate to elevate metabolism (Saint Girons et al. 1980), and thus closed forest canopies which minimise forest floor light patches may influence the activity, and hence the hunting success of tuatara. In contrast, the lower canopy height and less developed leaf litter of early regenerating forest may provide sub-optimal levels of invertebrate foods and decreased water retention of forest soils and litter, exacerbating the susceptibility of tuatara to cutaneous water loss (L. Hill (pers. comm.) in Barwick 1982).

The availability of underground refuges (e.g. rock crevices, burrows) for tuatara may also affect their ability to successfully establish new populations. As adults, tuatara appear extremely site attached with a small (ca. 10-20m radius) territory size (Newman & McFadden submitted, pers. obs.), but may range much greater distances from core refuges (ca. 100m pers. obs.) during the courtship season. Little is known about how mates find each other, but courtship appears to be conducted using only visual stimuli (Cree & Thompson 1988). Tuatara will dig their own burrows (over a period of days or weeks) but most tuatara on islands use seabird burrows. If burrow availability determines the degree of dispersal amongst released tuatara, then individuals re-introduced to low density burrow areas may disperse widely and become site attached far away from potential mates, with a corresponding reduction in reproductive output of founders.

Hypotheses such as these can easily be incorporated into re-introduction programmes to test the importance of individuals environmental factors. The recent re-introduction of tuatara to Moutohora in 1996 provided the opportunity to experimentally test the effects of vegetation type and burrow availability on the dispersal, survivorship and condition of tuatara. Animals were released into areas of open and closed canopy forest as well as areas supporting large numbers of evenly distributed seabird burrows and areas with aggregations of burrows interspersed with no burrows. If canopy closure and age of forest affected the ability of tuatara to hunt, feed and maintain body condition, then tuatara released into closed, early successional forest sites were expected to lose body condition at a greater rate than those released into more open canopy, mature forest sites. The results of that study are presented in depth in Chapter 3, but, generally, found no significant effect by vegetation type within release areas on tuatara condition or survivorship, but did note a significant effect on dispersal patterns related to the dispersion of seabird burrows.

Linking trials of techniques or methods with species re-introductions can also provide valuable information to guide future re-introductions. Trials of artificial burrows on Moutohora indicate that tuatara will use artificial refuges. This indicates that islands without natural seabird populations,
whose restoration could take many decades, may be viable re-introduction sites if artificial burrows can prevent excessive dispersal of founders from release areas. Well-planned comparisons of dispersal of tuatara from release sites with and without artificial burrows as the only refuges (e.g. Tiritiri Matangi tuatara proposal Chapter 5) will better assess the potential of this management tool.

Public access to tuatara is seen as a management priority (Cree & Butler 1993), but there is concern about impacts of people on newly released tuatara populations, especially from illegal wildlife traders. Carefully designed releases will provide valuable information on the impacts of public access on tuatara and vice versa, as well as quantifying the threat that wildlife traders pose to publicly accessible tuatara populations. Likewise, current management issues such as the optimal temperature regime for incubating captive raised eggs of remnant populations, and the merits of captive-reared or wild caught juvenile or adult tuatara as founders for new populations, need to be tested as well-planned experiments so that reliable information can be generated to guide management. Studies of the fitness of captive-raised offspring raised under a range of incubation temperatures are already under way (Nicola Nelson pers. comm.).

CONSERVATION IN A CHANGING WORLD - WHERE CAN HYPOTHESIS TESTING HELP?

The following section discusses the assumptions underlying some of the most common conservation tools or emerging conservation issues. Factors from each of these areas which need testing are listed (Table 5) along with suggested criteria for evaluating the success of alternative conservation options. Criteria for choosing new sites or evaluating populations in current sites need not be always dominated by requirements for biological sustainability or for maximising reproductive output. Where conservation projects have primarily non-biological objectives (e.g. economic or advocacy), criteria for success should also emphasis these values. In some cases, this may mean that managers accept marginal rates of recruitment into populations and intensive management is required to ensure the survival of founders.

The list of experimental alternatives given below (Table 5) is by no means complete. Effective conservation alternatives to currently accepted practices will require constant refinement and addition to the alternatives given here and their associated criteria for measuring success. Most important however, is the need for transparent processes in the selection of final evaluation criteria so that criteria set are effective and represent the views of all stakeholders in conservation.
Modelling and representativeness

A range of population modelling and comparative habitat analyses are available to conservation biologists. Population Viability Analyses (PVA) are used to predict the growth rates and persistence of re-introduced populations (Lovegrove 1992, Lindenmayer 1994) and to assess the harvesting potential of populations as sources for re-introductions (Hoyle 1993). Bio-climatic modelling can be used to predict the occurrence of species within similar climatic conditions and requires a minimum of input information (Lindenmayer et al 1991, Lindenmayer 1994). Likewise, the advent of cartographic and analytical tools such as Geographic Information Systems (GIS) can provide planners and conservation biologists with tools for spatial landscape analysis (Beatley 1994).

Table 5. Conservation issues in species recovery requiring experimental testing and suggested evaluation criteria for measuring the success of these alternatives.

<table>
<thead>
<tr>
<th>Conservation issue</th>
<th>Factor requiring testing</th>
<th>Suggested evaluation criteria</th>
</tr>
</thead>
<tbody>
<tr>
<td>Modelling and representativeness</td>
<td>can species establish in different habitats</td>
<td>-long-term biological persistence</td>
</tr>
<tr>
<td>Habitat alternatives</td>
<td>urban vs non-urban habitats</td>
<td>-enhanced public support</td>
</tr>
<tr>
<td></td>
<td>presence vs absence of people</td>
<td>-economic self-sustainability or surplus</td>
</tr>
<tr>
<td></td>
<td>island vs mainland habitats</td>
<td>-marginal biological establishment</td>
</tr>
<tr>
<td></td>
<td>predator exclusion sites vs predator control sites</td>
<td>-enhanced public support</td>
</tr>
<tr>
<td></td>
<td>tested vs untested habitat types</td>
<td>-survivorship of founders</td>
</tr>
<tr>
<td>Climate change</td>
<td>within vs outside current biogeographic range</td>
<td>-biological self-sustainability</td>
</tr>
<tr>
<td></td>
<td>manage in current vs new habitat types</td>
<td>-maximal productivity</td>
</tr>
<tr>
<td></td>
<td></td>
<td>-cost-effectiveness</td>
</tr>
<tr>
<td></td>
<td></td>
<td>-marginal biological establishment</td>
</tr>
<tr>
<td></td>
<td></td>
<td>-short-term survival and breeding</td>
</tr>
</tbody>
</table>
These procedures make critical assumptions of habitat suitability or predict population demographics based on data from existing habitats which may not be representative of new re-introduction habitats. Consequently, predictions from such models for experimental re-introductions should be treated with caution. For example, the programme BIOCLIM (Nix 1986), a bio-climatic modelling procedure, requires only longitudinal, latitudinal and altitudinal data from remnant populations to make predictions of habitat suitability. Using BIOCLIM, morphological and behavioural traits recorded for takae in alpine grassland refuges would merely reinforce propositions of takae as a purely alpine species, rather than revealing alternative lowland habitats and more benign climates (see previous).

Similar problems exist with PVA models. The reliability of predictions is highly dependant on the quality of information fed into models. Potential re-introduction sites may be unjustifiably omitted from management plans on the biological basis of habitat size, presence of undesirable factors (e.g. predators) or perceived intrinsic population growth rates. Territory size is often not consistent between habitats (e.g. kiwi cf. Colbourne & Kleinpaste (1963) vs. Potter (1989)) and is known to vary greatly among individuals even in the same habitat (e.g. McLennan & McCann 1991, Giradet unpubl.). Therefore, predictions of carrying capacity and long-term stability under genetic, demographic and environmental stochasticity (May 1991) for alternative re-introduction sites may be either overly pessimistic or optimistic. Intrinsic growth rates for populations in alternative habitats may exceed those recorded in source habitats, therefore decreasing predicted risk of extinction from environmental catastrophe (Lawton 1997) or increasing long-term population viability in the presence of competitors or predators.

Meaningful application of population modelling and habitat identification will only be possible if reliable information on species biology and habitat needs is available. Carefully designed experimental re-introductions are the logical means for determining the likely variability in this information (Table 5) and thus increasing the reality of such analytical procedures.

**Public access to biological heritage; conservation in modified or novel environments**

Far more habitat options exist for threatened species than are represented by their remnant distributions. The potential for restoration projects in alternative environments will only ever be known if the full range of options, including urban mainland sites, are tested through experimental comparisons (Table 5). Surveys demonstrate that the general public want a greater say in conservation and greater access to natural heritage (Craig & Stewart 1994, Craig et al. 1995). The
role that an informed, educated and supportive public plays in enabling species recovery is recognised as a key objective of New Zealand national conservation policy documents such as the Draft Biodiversity Strategy (DoC - MfE 1998) and the Department of Conservation's annual strategic business plans (e.g. DoC 1998). Yet in New Zealand opportunities to view native wildlife in natural settings are severely limited by the remote locations (mainland or offshore islands) or fragile environments that remnant populations inhabit.

Restoration projects near population centres are few, but increasing, and are limited mainly to inshore islands. They cannot by themselves provide for either the increasing popularity of such sites for visitors or for the wide range of flora and fauna that could be re-introduced to restored ecosystems. Indeed, the popularity of New Zealand's first public access wildlife sanctuary (Tiritiri Matangi Island) has meant that limits to the daily number of visitors have to be placed to prevent erosion of personal experience (DoC 1995). Moreover, surveys of visitors show that the majority are high income earners (Craig & Stewart pers. comm.), and it is probable that the high travel costs incurred by visiting the sanctuary prevent effective representation by all socio-economic groups.

The positive experiences of visitors to wildlife sanctuaries such as Tiritiri also demonstrate that while public education and wildlife advocacy are important, the most effective way to foster interest and support for conservation is to encourage direct access to wildlife. Re-prioritisation of conservation goals to emphasise re-introduction of threatened species to public population centres has been suggested as one method of better linking people with their natural heritage (Craig 1997). It is encouraging to see that preliminary planning for the revision of the Tuatara Recovery Plan (DoC 1999) includes the restoration of tuatara to its historic range (including mainland areas) as the long-term goal and the identification of controlled public access to tuatara in the wild as one of four short-term objectives. Only through bold experimentation with alternative recovery options will the full value of these habitats and approaches to species recovery planning be determined.

Criteria for establishing new populations of species should reflect the site-specific gains from undertaking the re-introduction (Table 5). For example re-introductions to areas where conservation values outweigh public access values, and for which ongoing management must be minimised (due to personnel or financial costs) should establish site selection criteria which demand a near or fully self-sustaining population without management intervention as the objective (e.g. island populations). By comparison, re-introduction of species to highly modified or novel habitats (such as urban settings) may include provisions for ongoing and intensive management to ensure a sustainable population is reached (e.g. supplementary feeding, intensive predator control).
Global climate change

Average global temperature is predicted to rise between 1.5-4.5°C by ca. 2050 (Inter-governmental Panel on Climate Change 1990), an increase of 10-60 times the natural rate of temperature change (Schneider 1989). Estimates for the maximum rate of migration for many plants and animals suggest that habitat change and loss from global warming will severely decrease current ranges and, in many cases, result in local or total extinction of species without human intervention (Dennis & Shreeve 1991, Lindenmayer et al. 1991, Huges & Westoby 1994). Strategies for mitigating impact have largely focused on conservation of native plant corridors to enable migration from habitat fragments (Hobbs & Hopkins 1991) or planting of native plants outside their current ranges in anticipation of their need as both genetic reservoirs for plants and viable habitat for fauna (Ledig & Kitzmiller 1992, Huges & Westoby 1994).

However, little attention has been given to the important role that experimentation with viable habitats for fauna have for alleviating the effects of global climate change. Threatened species may possess greater plasticity in their use of habitats and tolerance of temperatures dissimilar to their current refuge areas than is generally thought. Evaluating habitat use in dissimilar environments may identify alternative, viable habitats or identify which native species can be planted to ensure minimal time for habitat creation and production of habitat needs (such as shelter and food), and help prioritise species which need assistance in migrating to more favourable locations.

CONCLUSIONS

Decisions for the management of threatened species carry risks, whether they involve restoration of species to areas similar to those where outcomes have been beneficial, or to new areas where the response of species is unknown. Increased efficiency and effectiveness of species management will only occur if the risks in undertaking actions can be quantified against alternatives and the benefits of undertaking novel management approaches can be explicitly demonstrated. Undertaking restoration projects, such as re-introductions, as trials may allow refinement of new methodologies but do not allow the reasons for success or failure to be identified with certainty. Therefore, the reliability of repeating specific management actions is unknown and success in one habitat type cannot be duplicated within similar or dissimilar habitats with confidence.

Incorporating scientific principles into project design offers greater certainty for identifying why projects succeed or fail. However, the initial investment costs may be higher than for trials (although they reduce over successive projects) in terms of money, labour and the number of replicate sites and study individuals required. Basic ecological assumptions underlie most management actions for
threatened species recovery such as re-introduction biology. Testing these assumptions as management alternatives, and as well-planned experimental re-introductions underpinned by scientific principles, will help identify the habitat plasticity of species and the range of options available to managers to assist recovery. Additionally, testing experimental alternatives to current management approaches will help refine decisions made by managers such as for the selection of re-introduction sites and the degree of management action required (e.g. pest control) to effect recovery.

Adopting a more rigorous scientific approach to re-introduction biology is necessary both for its emergence as a science and for its effectiveness in resolving future challenges to biodiversity preservation such as integrating people into conservation and mitigating the effects of global climate change.

ACKNOWLEDGEMENTS

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TUATARA
Sphenodon spp.

Reproduced from Powell, A. W. B. 1993
Native animals of New Zealand (revised edition)
Auckland Museum & Institute, NZ

NEW ZEALAND

KIORE
Rattus exulans

Plate 2
Chicken Island group showing locations of study sites for assessing dietary interactions between tuatara and kiore
DIETARY INTERACTIONS BETWEEN TUATARA AND KIORE: IMPLICATIONS FOR RECOVERY AND MANAGEMENT

Abstract

Compelling circumstantial evidence links the introduced Pacific rat or kiore (*Rattus exulans*) to the decline of New Zealand's endemic reptile, the tuatara (*Sphenodon spp.*), but direct experimental evidence is lacking. The eradication of kiore from the tuatara-inhabited Chicken Islands, northern New Zealand, has offered the first opportunity to experimentally investigate one component of potential competition - the existence and intensity of inter-specific food competition. The diets of adult tuatara were compared before and after kiore removal from regenerating and mature forests during summer and autumn of 1993-1997 on two of these islands.

Kiore successfully out-competed adult tuatara for food items but the intensity of competition was not equal between seasons or forest types. A significantly greater proportion of tuatara had food in their stomachs in both habitats after kiore removal, but the number and size of prey eaten increased significantly only for tuatara from regenerating forest, not for those from mature forest. Adult tuatara from regenerating forest also demonstrated a consistently greater contraction in dietary breadth than tuatara from mature forest and had the greater level of prey switching following kiore removal.

1 Submitted to *Pacific Conservation Biology* January 1999
Moreover, responses in dietary breadth and prey switching to the removal of kiore were more pronounced during autumn compared to summer for both habitats. Overall, the intensity of food competition was consistently greater in the regenerating forest than the more mature forest, although interference competition (the ability of tuatara to successfully catch prey) was present in both habitats.

This study supports hypotheses which propose kiore as the major factor leading to the decline or extinction of tuatara populations. However, the results also support models which propose that the impact of kiore on wildlife is related to the level of historic, human-induced, habitat modification. Eradication of kiore is currently the only management solution for restoring native wildlife, but the emerging concerns of some scientists and Maori tribes encourage the investigation of alternative approaches to the management of kiore-tuatara interactions. To achieve this, management actions would benefit from rigorous hypothesis testing of cause and effect and reflect the quality of information currently available.

Key words: tuatara; kiore; Pacific rat; Sphenodon; Rattus exulans; species recovery, co-management

INTRODUCTION

Although once widespread on mainland New Zealand in pre-human times (e.g. Worthy 1995; Worthy & Holdaway 1997), New Zealand's endemic tuatara (Order Sphenodontida; Sphenodon spp.) is now restricted in natural populations solely on offshore islands. The IUCN Red Data Book lists tuatara as 'endangered' (Brothers Island tuatara S. guntheri), or 'rare' (Northern tuatara S. punctatus) (Baille & Groombridge 1996). Despite an historically high level of scientific interest and its current conservation status, there remains heated debate over what factors threaten tuatara populations and the exact actions necessary to guarantee their future viability.

Central to this debate is the role of the introduced Pacific rat or kiore Rattus exulans which is present (or was until recently) on 9 of the 32 offshore island refuges for Northern tuatara (Cree & Butler 1993). The human history of environmental destruction on New Zealand offshore islands is similar to that of the New Zealand mainland, with forest clearance and introduction of exotic mammals firstly by Polynesian settlers (approx. 1000 BP) and later by Europeans (200 BP) (Davidson 1990; Hayward 1986). Polynesian settlers brought with them the kiore and the Polynesian dog (kuri), and European settlers introduced an additional three rodent species, mustelids, cats (Felis felis), dogs (Canis familiaris) and a wide range of other mammals. The disappearance of tuatara from mainland
New Zealand before the advent of Europeans is variously attributed to human-induced habitat destruction and the presence of kiore and kuri (e.g. Towns & Daugherty 1994; Whitaker 1978), although the relative importance of each of these factors is unknown. Recent designation of many important island sanctuaries as wildlife reserves has led to the cessation of farming activities and removal of feral stock which in turn has enabled several severely degraded islands to regain forest cover (e.g. Bellingham 1984) and, superficially, some resemblance of their original state.

Clear evidence of the impact of the Norway rat (Rattus norvegicus) has been demonstrated by its extermination of tuatara from Whenuakura I. between 1981 and 1984 (Newman 1986, 1987). There is, however, no certain evidence for a similar impact by kiore on tuatara. Instead, circumstantial evidence for the nature and degree of impact is inferred from indirect studies of tuatara and kiore interactions. These include the apparent absence of juvenile tuatara from kiore-inhabited islands (Cree et al. 1995; Crook 1973), smaller clutch sizes of tuatara on islands with kiore (Newman et al. 1994), the probable predation by kiore on one juvenile tuatara (Newman 1988), and the considerable overlap which exists between the diets of kiore and tuatara (Newman & McFadden 1990; Walls 1981; Ussher 1999, unpubl. data). Also, most kiore-inhabited tuatara islands also support low densities of adult tuatara and, unlike kiore-free islands, few juvenile individuals have been found (Cree et al. 1995). However, although these data generate plausible hypotheses for mechanisms of impact, investigations based on direct observation of egg and juvenile predation or inter-specific food competition have yet to be undertaken.

Untested hypotheses of kiore impact are used as a basis for implementing eradication programmes for kiore from islands (e.g. Cree & Butler 1993). This has raised concerns about the quality of science guiding management and the way in which investigations of cause and effect relationships between kiore and native fauna are approached (Craig 1986). Craig (1986) also calls for models of tuatara decline to incorporate aspects of each island's history, including factors such as past forest clearance and the presence of other exotic mammals, in addition to kiore.

Understanding the role of kiore in the decline of tuatara (and other fauna) is becoming increasingly important as some Maori seek recognition in management of kiore as a culturally significant species (Roberts 1995; Hori Parata pers. comm.) and conservation biologists call for a more balanced and scientific investigation of possible impacts (Craig 1986; Cree et al. 1995; Veltman 1996). A major force shaping debate over the management of kiore is the difference in cultural value attributed to kiore by some Maori compared to the predominantly European-based management authority - the Department of Conservation (DoC) (Craig et al. 1995). Roberts (1995) contrasts the close association between rats and disease for European descendants to the reliance by Maori settlers on kiore as an important food source and indication of status and wealth. Indeed, kiore has been
incorporated into every aspect of Maori culture including cosmology and genealogy, trade, songs, carvings, names and place-names (Haami 1992). *Rattus exulans* is the third most widely distributed rodent in the world (Wodzicki & Taylor 1984), and occurs on approximately 40 New Zealand islands or island groups, other than those also supporting tuatara, either as the only introduced mammal or in conjunction with other introduced rodents and mammals (Atkinson & Moller 1990; DoC 1994). However, most islands with tuatara have a long history of Maori settlement and resource use and the conservation of both kiore and the native biota on them are regarded as equally important goals for some regional tribes (Hori Parata pers. comm.).

The recent eradication of kiore from the Chicken Islands by DoC offers the opportunity to test hypotheses of kiore impact on tuatara. Female tuatara reproduce only every 4 or 5 years (Newman *et al.* 1994), nests are well hidden and young tuatara are extremely cryptic and difficult to survey. Therefore, changes in nesting success as well as egg and hatchling survivorship are difficult to monitor. Competition by kiore for food may also be important in affecting the breeding condition of tuatara and therefore indirectly influence the rate of juvenile recruitment (c.f. predation of eggs or young) into populations.

In contrast to these difficulties, the responses of tuatara to release from inter-specific food competition can be measured immediately, and easily, following the removal of kiore. A preliminary assessment of the potential for tuatara and kiore to compete for food has already been undertaken on the Chicken Is. That study showed that adult tuatara in the presence of kiore ate proportionally less large (>10mm in length) invertebrate prey (Ussher 1999) than did adult tuatara on islands without kiore (Walls 1981; Fraser 1993). Moreover, kiore in the presence of tuatara ate the same types of invertebrate foods from the same leaf litter habitats that tuatara fed from (Ussher 1995). Kiore also ate invertebrate prey ranging in size from 3mm to greater than 20mm in length (Ussher 1995), suggesting that kiore also compete for food with hatchling and juvenile tuatara which, from studies on islands without kiore (Fraser 1993), eat smaller (<10mm) prey than adult tuatara. Kiore are also presumably more efficient hunters of invertebrate prey than tuatara. Rodents such as kiore actively search for food, detecting prey using smell and sight. Tuatara employ a passive hunting strategy, remaining motionless at one location and using contrast between prey and habitat to detect potential foods (Walls 1981; Meyer-Rochow & Teh 1991). The intensity of competition between kiore and tuatara will be influenced by the numbers of each species present. Studies show that population cycles, peak annual density and annual productivity of kiore populations differ between forest types (Bunn 1979; Craig 1986; Bunn & Craig 1989; Roberts & Craig 1990). Further, studies suggest that the magnitude and duration of annual fluctuations of kiore numbers decrease as forests mature following disturbance (Craig 1986; Roberts & Craig 1990). If the numbers of kiore present are
related to the degree of intensity of food competition with tuatara, then the diets of tuatara in more mature forest should change less than those in younger forests.

This study aims to compare the diets of tuatara before and after the removal of kiore in order to test the hypothesis that kiore out-compete adult tuatara for food. Additionally, to test hypotheses of reduced competition in younger forests, dietary responses of adult tuatara were compared between regenerating (young) and mature forests. Removal of an efficient invertebrate predator such as kiore should result in immediate increases in the availability of invertebrates. Also, natural fluctuations in the availability of foods for tuatara could influence the ability of tuatara to feed. Therefore, invertebrate populations were monitored in one of the habitats to test both predictions of increased food availability following kiore removal and to distinguish between changes in diets due to increased access to similar numbers of invertebrates or increases in invertebrate populations.

**STUDY AREA AND METHODS**

Research took place on Lady Alice I. (138 ha) and Whatupuke I. (90 ha) in the Hen and Chickens Is. Group (35° 50'S, 174° 45'E) (DoLS 1976; Plate 2), Northern New Zealand. Both islands support large populations of tuatara and, until recently, kiore (Crook 1973; Newman & McFadden 1990). Mature forest removed by Maori and European activities (Hayward & McCallum 1984, Prickett 1984), has largely regenerated with both islands now supporting distinctive areas of late successional mixed broadleaf forest (mature forest) and earlier mid-successional kanuka *Kunzea ericoides* forest (regenerating forest) (Percy 1956; Ritchie & Ritchie 1970). There are no records of exotic predators or competitors, other than kiore, being present on these islands in the past. Cattle were present on Lady Alice I. for 34 years but were removed by 1924 when farming ceased on the islands (Hayward & McCallum 1984). There is no written record of the time of arrival of kiore on the Chicken Is. However, Maori oral tradition tells of established annual harvests for kiore up until approximately 100 years before present (Hori Parata, pers. comm.). Kiore were harvested from these islands up until they were abandoned by Maori in the late 1800's although the scale and frequency of these harvests is unknown (Hori Parata pers. comm.).

Kiore were removed from Whatupuke I. in October 1993 and from Lady Alice I. in September 1994. Both eradications were instantaneous aerial poison campaigns (c.f. the longer term ground campaign on Coppermine Island, another island of the Chickens Group). The majority of kiore from each population are considered to have been killed within 3-4 days of the poison drop (Ray Pierce pers. comm.). Intensive post-eradication monitoring by DoC found no sign of kiore after 1-2 weeks of the poison drops (Ray Pierce pers. comm.). The earliest sampling period for this study following
each of these eradications was approximately 4 months. No previous eradication had been attempted on either island.

Table 1. Timing of tuatara diet and invertebrate sampling plus the timing of eradication of kiore in the three study sites during the project. ✓ = kiore present; ✗ = kiore absent; - = not sampled.

<table>
<thead>
<tr>
<th>Sampling period</th>
<th>Lady Alice I. regenerating forest</th>
<th>Lady Alice I. mature forest</th>
<th>Whatupuke I. mature forest</th>
</tr>
</thead>
<tbody>
<tr>
<td>Summer 1993</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Autumn 1993</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
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<tr>
<td>Summer 1994</td>
<td>✓</td>
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<td>Autumn 1994</td>
<td>✓</td>
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<td>Spring 1994</td>
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<tr>
<td>Autumn 1997</td>
<td>✗</td>
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</tr>
</tbody>
</table>

Changes in the diet of adult tuatara (hereafter referred to as 'tuatara' unless otherwise specified) following the removal of kiore were studied in one regenerating forest site and one mature forest site on Lady Alice I., and one mature forest site on Whatupuke I. (Table 1, Plate 2). Neither kiore nor tuatara densities were accurately measured in the sites. Observations over the study suggest that overall tuatara densities in both mature forest sites were approximately 30% less than those in the regenerating forest site. However, tuatara were not evenly distributed in each habitat and individuals were sampled from areas in each habitat where local densities were similar between study sites (areas of high and low local density). Density and dispersion of tuatara appeared similar between mature sites on both islands. Juvenile tuatara were rarely seen during this study. Two post-hatchling sized tuatara (approx. 1 year old) as well as 3 juvenile or sub-adult animals were found in the Lady Alice I. regenerating site, and two sub-adult individuals in each of the mature forest sites were seen prior to kiore removal.
Sampling was restricted to summer and autumn seasons, because kiore abundance (Craig & Bunn 1989) and tuatara activity (Walls 1983) peak during these months. This implies that the intensity of competition between the two species will be greatest at this time. A single spring sample of tuatara diets and environmental invertebrate availability was taken in 1994. Both islands were sampled over two week periods within each season and at similar times for consecutive years. In Autumn 1997, only the regenerating habitat on Lady Alice I. could be sampled.

**Invertebrate availability**

Abundance and diversity of invertebrates was estimated simultaneously with tuatara diets using pan-traps to sample litter dwelling invertebrates. Pan-traps consisted of a clear plastic dish (approximately 24 x 16 x 4.5 cm) placed level with the top of the leaf litter and filled ⅓ full with a mixture of water and detergent (to ensure removal of water surface tension). The upper lip of each trap was made level with the leaf litter by packing the external sides of each trap with soil and leaf litter. Twenty traps were set at 10m intervals in each respective habitat, emptied every 24 hours and set for periods of 4-5 days on each trip. Invertebrate populations were monitored only in mature forest sites.

**Classification of invertebrates**

Classification and analysis of tuatara dietary items and invertebrates sampled from the environment reflected both rate of inclusion in diets and differences in habit. Diets of tuatara sympatric with kiore on the Chickens Is. consist mainly (74%) of insect prey (Ussher submitted). Therefore, most non-insect species were grouped at Class taxon while most insects were identified to Order taxon. Exceptions were: the division of Class Malacostraca into Order Amphipoda and Order Isopoda based on differences in locomotion and therefore, presumed vulnerability of members to predation by tuatara; insect larvae (Order Lepidoptera, Order Coleoptera and Order Diptera) which were considered as one prey group; and weta species which were divided according to habitat as ground-dwelling (Subfamily Rhaphidophorinae and Subfamily Stenopelmatidae) or tree dwelling (Subfamily Stenopelmatidae) respectively.

**Diets of tuatara**

Adult tuatara used in this study ranged in size from 183 - 250mm snout-vent length (SVL) for females and 181 - 272mm SVL for males. Repeat sampling of individuals within visits was avoided by marking animals which had been sampled. Repeat sampling of individuals between visits was not avoided, because intervals between visits (typically 2-3 months) were considered great enough for
any effects of previous sampling to disappear. Tuatara typically shelter in underground burrows (usually seabird burrows) by day and emerge at night to feed. Adult tuatara active on the forest floor were caught by methodically searching the same areas of habitat on each visit. Each individual caught was stomach-flushed using a modified pressure pump (see Legler & Sullivan 1979 for basic technique; Carmichael et al. 1989 for adaptation to tuatara and Fraser 1993 for extensive application to tuatara), and the stomach contents preserved in alcohol for later identification. Prey items were examined and identified under a stereo-microscope, and size of whole animals was estimated using an invertebrate reference collection from the Chicken Is. Minimum number of prey items was estimated by counting body parts present; for example six legs or two similar antennae constituted a single insect. Items were classed according to body length and were placed into either >10 mm or 3-10 mm size classes. Previous study of the diets of adult tuatara on the Chicken I. indicates that prey smaller than 3mm in length are not eaten (Ussher 1999).

Analysis

Aspects of the data which were analysed were: sampling coverage (see below); number of prey eaten; size of prey eaten; foraging success (% of tuatara with food in their stomach); dietary breadth (diversity and proportion of prey groups eaten); and dietary overlap (similarity between the diets of tuatara before and after kiore removal).

The efficiency of sampling for estimating the proportions of prey groups or number of prey groups consumed by tuatara was calculated from Fagen and Goldman's estimate of sampling coverage (Lehner 1979) where \(1.0 = \text{total coverage and } <1.0 = \text{reduced coverage}\) (see also Clode & MacDonald 1995). Only those invertebrates likely to be food items of tuatara (from studies by Walls 1981; Ussher 1999) were included in analyses of tuatara feeding ecology. These were overwhelmingly ground-dwelling species. Invertebrates selected for inclusion comprised between 86% and 97% of all dietary items appearing in tuatara diets for this study. Incomplete monitoring of invertebrate availability and small sample sizes from Whatupuke I. in 1993 prevented comparison of some aspects of diet with animals on Lady Alice I.

Number of prey consumed by tuatara were log-transformed (to counter non-normality) before analysis using Analysis of Variance (ANOVA). None of the interactive effects were significant and were therefore removed from models. Because invertebrate availability varies with season (e.g. Moeed & Meads 1985), models for foraging and size-related feeding responses (PC SAS (SAS 1989) categorical data analysis (CATMOD); maximum likelihood Analysis of Variance) tested kiore presence and also seasonal variables (replicate seasons only - spring sample not included in
analysis). Interactive effects of kiore eradication and season (treatment x season) were non-significant for most models, allowing removal of this interaction following preliminary analysis. Significant interactive effects warranted separate testing of 'kiore removal' as a variable, within seasons. Zero values in data-sets were substituted by 0.001 during analysis, meaning that differences in final p-values between approximations and actual zero values were at least one order of magnitude less than the minimum alpha level of significance set for all analyses (0.05). Because most hypotheses following removal of kiore predict increases in aspects of tuatara diets, most statistical tests used were one-tailed. For changes in the frequency of occurrence of specific dietary items, two-tailed tests were applied to detect both increases and decreases in diets. Dietary breadth and overlap were calculated from Levins' formula (Krebs 1989), and Pianka's (1976) adaptation of MacArthur and Levins' formula respectively as applied in Clode and MacDonald (1995).

RESULTS

The numbers of tuatara sampled were sufficient to ensure that most prey groups eaten by tuatara were recorded. Coverage of prey groups by sampling was high in both habitats both before and after kiore removal (kanuka: before 0.95, after 0.98; broadleaf: before 0.87, after 0.97) suggesting that increasing the number of samples would not greatly increase the numbers or proportions of prey groups recorded.

Invertebrate abundance varied between seasons and years but overall there was no consistent change in either total invertebrate abundance or large invertebrate abundance in mature forest following kiore removal from Lady Alice l. (Fig. 1b). There is a similar pattern on Whatupuke l. over different years in the absence of kiore (Fig. 1c).

Number of items consumed and prey size

There was a marked difference between habitats in the number of prey in the stomachs of tuatara following kiore removal. Tuatara in regenerating forest consumed significantly more prey after kiore were removed (ANOVA df(1, 91), F=5.39 p<0.05), but those in mature forest ate similar amounts before and after kiore were removed (ANOVA df(1, 71), F=0.45, p>0.5). Tuatara in regenerating forest ate fewer prey than tuatara from mature forest in both summer and autumn when kiore were present (Table 2). However, after kiore were removed, numbers of prey eaten by tuatara from regenerating
regenerating forest, Lady Alice l.

mature forest, Lady Alice l.

mature forest, Whatupuke l.

Season (summer, autumn, spring) and Year

Figure 1. Total invertebrate abundance, abundance of large invertebrates and foraging success (% of individuals with food in stomach) of tuatara on Lady Alice (1a: regenerating forest; 1b: mature forest) and Whatupuke (1c: mature forest) Is. before and after the removal of kiore. Arrows denote timing of kiore eradication. Sample sizes for tuatara stomach analyses are shown above columns.
forest rose to more closely resemble numbers eaten by tuatara in mature forest. None of the pairwise comparisons of change in number of prey eaten with the removal of kiore, nor comparisons between numbers of prey eaten between sites either before or after kiore removal, was statistically significant.

Table 2. Mean (+/− S.E.) number of prey eaten by tuatara in regenerating and mature forest before and after the removal of kiore.

<table>
<thead>
<tr>
<th>SITE</th>
<th>KIORE PRESENT</th>
<th></th>
<th>KIORE ABSENT</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Summer</td>
<td>Autumn</td>
<td>Summer</td>
<td>Autumn</td>
</tr>
<tr>
<td>Regenerating forest</td>
<td>1.8 (0.3)</td>
<td>2.0 (0.7)</td>
<td>2.7 (0.3)</td>
<td>3.4 (1.6)</td>
</tr>
<tr>
<td>Mature forest</td>
<td>2.9 (0.3)</td>
<td>3.2 (0.4)</td>
<td>4.1 (0.8)</td>
<td>3.2 (0.5)</td>
</tr>
</tbody>
</table>

The removal of kiore also resulted in a significantly greater increase in the proportion of large prey eaten by tuatara in regenerating forest (total diets: 70% n=26 before vs. 86% n=88 after kiore removal), but not for tuatara in mature forest (total diets: 78% n=24 before vs. 83% n=50 after kiore removal; Table 3a). There was no effect of season.

Table 3. Increase in proportion of large (>10mm) invertebrate prey consumed (3a) and foraging success (% of individuals with food in their stomach) of tuatara (3b) following kiore removal from two habitats on Lady Alice I. SIG = significant, NS = not significant.

3a: Large (>10mm) prey consumption.

<table>
<thead>
<tr>
<th>Habitat</th>
<th>Effect</th>
<th>Df</th>
<th>Chi-Square</th>
<th>Probability</th>
</tr>
</thead>
<tbody>
<tr>
<td>Regenerating forest</td>
<td>Kiore removal</td>
<td>1</td>
<td>6.45</td>
<td>&lt;0.05 SIG</td>
</tr>
<tr>
<td></td>
<td>Season</td>
<td>1</td>
<td>3.33</td>
<td>0.07 NS</td>
</tr>
<tr>
<td>Mature forest</td>
<td>Kiore removal</td>
<td>1</td>
<td>1.32</td>
<td>0.25 NS</td>
</tr>
<tr>
<td></td>
<td>Season</td>
<td>1</td>
<td>2.01</td>
<td>0.16 NS</td>
</tr>
</tbody>
</table>

3b: Foraging Success

<table>
<thead>
<tr>
<th>Habitat</th>
<th>Effect</th>
<th>Df</th>
<th>Chi-Square</th>
<th>Probability</th>
</tr>
</thead>
<tbody>
<tr>
<td>Regenerating forest</td>
<td>Kiore removal</td>
<td>1</td>
<td>20.49</td>
<td>&lt;0.001 SIG</td>
</tr>
<tr>
<td></td>
<td>Season</td>
<td>1</td>
<td>0.05</td>
<td>0.83 NS</td>
</tr>
<tr>
<td>Mature forest</td>
<td>Kiore removal</td>
<td>1</td>
<td>10.64</td>
<td>&lt;0.05 SIG</td>
</tr>
<tr>
<td></td>
<td>Season</td>
<td>1</td>
<td>2.57</td>
<td>0.11 NS</td>
</tr>
</tbody>
</table>
Foraging success

Foraging success of tuatara (% of individuals with food in stomach) showed a general increase (Fig 1a, b), which was statistically significant (Table 3b), in both Lady Alice I. habitats following the removal of kiore. Foraging success also increased following the removal of kiore from Whatupuke Island (Fig 1c) although reliability cannot be gauged due to the single sample for each season with kiore present. Foraging success for tuatara in mature forest showed no significant relationship to total invertebrate abundance either before (Spearman's Correlation Co-efficient $r^2=0.9$ n=4 $p>0.5$) (Fig 1b) or after (Spearman's $r^2=0.2$ n=4 $p>0.5$) kiore removal. Also, there was no relationship to changes in large invertebrate abundance either before ((Spearman's $r^2=0.0$) or after (Spearman’s $r^2=0.2$ n=4 $p>0.5$) kiore removal. By comparison, increasing foraging success on Whatupuke, while not significantly associated with total invertebrate abundance (Spearman’s $r^2=0.55$ n=8 $p>0.05$), was significantly related to increases in the abundance of large prey (Spearman’s $r^2=0.66$ n=8 $p<0.05$) (Fig. 1c).

Niche breadth and overlap

Dietary breadth (diversity and proportion of prey groups eaten) contracted markedly in regenerating forest for both summer and autumn seasons after kiore were removed, but not in mature forest (Fig. 2). In both habitats, diets of tuatara contracted more for autumn than for summer following kiore removal.

Dietary overlap (similarity between the diets of tuatara) was high for summer in both habitats (regenerating 0.91 and mature 0.85) indicating that the summer diets of tuatara scarcely changed once kiore were removed and that foraging focus remained largely within similar prey items. By comparison, the lower overlap experienced for autumn (regenerating 0.75 and mature 0.58) indicates that tuatara ate some/many different foods and changed the focus of their foraging in this season after kiore removal.

Thus, following the removal of kiore, tuatara in regenerating forest foraged from a subset of the previous dietary items in summer, but in autumn included new prey in their diets. In mature forest tuatara did not change their diet during summer after kiore removal, but during autumn included new prey and changed the frequency of capture of previous prey.
Prey switching and changing predation pressure

The proportion by which most (14 out of 21) invertebrate prey groups were eaten by tuatara did not significantly change following kiore removal (Appendix 2.1). Seven prey groups: arachnids; cockroaches (Order Blattaria); moths (Order Lepidoptera); ants/wasps (Order Hymenoptera); ground-dwelling weta and tree weta, were found in a higher proportion of stomachs although the increase was not consistent across habitats or seasons (Table 4). Tree weta in particular showed marked increases amongst diets, particularly in autumn when they had previously been absent (Fig. 3). This pattern was also repeated on Whatupuke Island where tree weta were also absent from autumn diets in the presence of kiore but appeared after kiore removal (chi-square=33.59 df=1 p<0.001). Wasps were the only prey group to show a significant decrease in occurrence amongst diets (Table 4).

<table>
<thead>
<tr>
<th>Habitat</th>
<th>Season</th>
<th>Dietary Breadth</th>
<th>% Change</th>
<th>Dietary Overlap</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Kiore Present</td>
<td>Kiore Absent</td>
<td></td>
</tr>
<tr>
<td>Regenerating</td>
<td>Summer</td>
<td>0.53</td>
<td>0.39</td>
<td>- 26%</td>
</tr>
<tr>
<td></td>
<td>Autumn</td>
<td>0.79</td>
<td>0.50</td>
<td>- 37%</td>
</tr>
<tr>
<td>Mature Forest</td>
<td>Summer</td>
<td>0.37</td>
<td>0.40</td>
<td>+ 7.5%</td>
</tr>
<tr>
<td></td>
<td>Autumn</td>
<td>0.48</td>
<td>0.41</td>
<td>- 15%</td>
</tr>
</tbody>
</table>

Figure 2. Change in dietary breadth (diversity and abundance of dietary components) and dietary overlap (similarity between tuatara diets) for tuatara on Lady Alice I. Circles represent relative dietary breadth (numerical values given above circles) and are to scale. N= sample size.
Table 4. Food groups which showed substantial change in frequency of occurrence (% presence/absence in tuatara stomachs) following kiore removal from Lady Alice I. mature forest. Statistics in bold used to infer significance of change. SIG = substantial change; NS = not significant. * = presence of an interactive effect between kiore removal and season requiring seasons within habitats to be analysed separately.

<table>
<thead>
<tr>
<th>Food Group</th>
<th>Forest type</th>
<th>Effect</th>
<th>df</th>
<th>Chi-Square</th>
<th>Probability</th>
<th>Direction of Diet Change</th>
</tr>
</thead>
<tbody>
<tr>
<td>Spiders/ Harvestmen (Class Arachnida)</td>
<td>Regenerating season</td>
<td>1</td>
<td>0.26</td>
<td>0.54 NS</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>0.26</td>
<td>0.81 NS</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Mature season</td>
<td>1</td>
<td>1.40</td>
<td>0.24 NS</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>74.33</td>
<td>&lt;0.001 SIG</td>
<td>Increase</td>
<td></td>
</tr>
<tr>
<td>Millipedes (Class Diplopoda)</td>
<td>Regenerating season</td>
<td>1</td>
<td>not</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>eaten</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>*Mature summer</td>
<td>1</td>
<td>35.77</td>
<td>&lt;0.001 SIG</td>
<td>Decrease</td>
<td></td>
</tr>
<tr>
<td></td>
<td>autumn</td>
<td>1</td>
<td>23.49</td>
<td>&lt;0.001 SIG</td>
<td>Decrease</td>
<td></td>
</tr>
<tr>
<td>Cockroaches (O. Blattaria)</td>
<td>Regenerating season</td>
<td>1</td>
<td>0.00</td>
<td>1.00 NS</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>0.30</td>
<td>0.58 NS</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Mature season</td>
<td>1</td>
<td>13.84</td>
<td>&lt;0.05 SIG</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>15.23</td>
<td>&lt;0.05 SIG</td>
<td>Increase</td>
<td></td>
</tr>
<tr>
<td>Moths (O. Lepidoptera)</td>
<td>*Regenerating summer</td>
<td>1</td>
<td>22.93</td>
<td>&lt;0.001 SIG</td>
<td>Increase</td>
<td></td>
</tr>
<tr>
<td></td>
<td>autumn</td>
<td>1</td>
<td>2.48</td>
<td>0.12 NS</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>*Mature summer</td>
<td>1</td>
<td>19.31</td>
<td>&lt;0.001 SIG</td>
<td>Increase</td>
<td></td>
</tr>
<tr>
<td></td>
<td>autumn</td>
<td>1</td>
<td>-</td>
<td>not eaten</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>Wasps/ Ants (O. Hymenoptera)</td>
<td>*Regenerating summer</td>
<td>1</td>
<td>30.69</td>
<td>&lt;0.001 SIG</td>
<td>Increase</td>
<td></td>
</tr>
<tr>
<td></td>
<td>autumn</td>
<td>1</td>
<td>1.77</td>
<td>0.18 NS</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>*Mature summer</td>
<td>1</td>
<td>0.19</td>
<td>0.67 NS</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>autumn</td>
<td>1</td>
<td>39.02</td>
<td>&lt;0.001 SIG</td>
<td>Decrease</td>
<td></td>
</tr>
<tr>
<td>Ground Dwelling Weta (SubF. Rhaphidophorinae/ Stenopelmatidae)</td>
<td>*Regenerating summer</td>
<td>1</td>
<td>78.29</td>
<td>&lt;0.001 SIG</td>
<td>Increase</td>
<td></td>
</tr>
<tr>
<td></td>
<td>autumn</td>
<td>1</td>
<td>1.47</td>
<td>0.23 NS</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>*Mature summer</td>
<td>1</td>
<td>0.21</td>
<td>0.65 NS</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>autumn</td>
<td>1</td>
<td>59.58</td>
<td>&lt;0.001 SIG</td>
<td>Increase</td>
<td></td>
</tr>
<tr>
<td>Tree Weta (SubF. Stenopelmatidae)</td>
<td>Regenerating season</td>
<td>1</td>
<td>0.46</td>
<td>0.50 NS</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>107.04</td>
<td>&lt;0.001 SIG</td>
<td>Increase</td>
<td></td>
</tr>
<tr>
<td></td>
<td>*Mature summer</td>
<td>1</td>
<td>1.51</td>
<td>0.23 NS</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>autumn</td>
<td>1</td>
<td>50.58</td>
<td>&lt;0.001 SIG</td>
<td>Increase</td>
<td></td>
</tr>
</tbody>
</table>
Figure 3.
Frequency of occurrence of tree weta among tuatara diets before and after the removal of kiore from Lady Alice Island. Arrows denote timing of kiore removal. Zero values = absence of tree weta in diets. - = not sampled. Number of diets are shown above columns.
DISCUSSION

Food competition between tuatara and kiore

Dietary overlap between sympatric species may be interpreted in two ways. Firstly, overlap may merely reflect the joint exploitation of the most available food and not necessarily imply competition (Colwell & Futuyama 1971). Alternatively, food competition may encourage marked changes in feeding behaviour by the less successful forager (Pianka 1976). When a competitor is introduced or removed, the intensity of competition can be measured by immediate dietary change, and the significance of the interaction, by changes in population persistence for the less successful forager.

Changes in the diets of tuatara recorded following kiore removal included increased rate of prey consumption, shifting focus towards larger prey, increased foraging success, narrowed predation focus, and prey switching - all indicators of increased prey availability. The consistency of results between Lady Alice and Whatupuake Islands strongly suggests that tuatara dietary responses are indeed due to kiore removal and are neither island-specific nor caused by confounding factors such as weather.

The general increases in foraging success by tuatara seen in both habitats need not be linked to food competition alone. Newman (1988) records the probable predation of a juvenile tuatara by kiore, and kiore are known to prey on skinks in the size range of juvenile tuatara (Newman & McFadden 1990; Ussher 1995). Anti-predator behaviour noted for invertebrates in the presence of rodents (Bremner et al. 1989) could also be also adopted by juvenile tuatara and skinks on kiore-inhabited islands, as was suggested by Craig (1986) based on observations by Dawbin (1982). Whilst it seems unlikely that kiore eat adult tuatara (because they co-exist on many islands), avoidance behaviours of juvenile tuatara may be expressed in adult animals enough to disrupt them from foraging when near kiore. Therefore, changes in foraging success in the presence of kiore could be related to both reduction in foraging time through interference competition and through the reduction of local prey abundance by kiore.

Changes in dietary breadth and overlap suggest both increased rates of predation on existing prey and the addition of new prey groups by tuatara once kiore were removed. One model of optimal foraging theory predicts greater specialisation by tuatara on prey most likely to generate the greatest net energy benefit following kiore removal (McArthur & Pianka 1966). Diets of allopatric adult tuatara reveal a preference towards tree weta (Walls 1981) (approximate length = 50-70mm, weight up to
14 grams (Moller 1985)) and large beetle species (approximate body length = 20mm (Ussher unpubl. data; Walls 1981)) when they are abundant. Diets of sympatric adult tuatara on Lady Alice Island comprised mostly insect larvae (approximate length = 10-20mm) and beetles (both large ~10mm and small 5-10mm) (Ussher 1999). The stark shift in the diets of tuatara from small (<10mm) prey towards larger weta following kiore removal may illustrate the degree to which kiore have successfully out-competed tuatara for these large foods. Unfortunately, changes in the abundance of tree weta following kiore removal are unknown because they are not effectively sampled by pan-traps. Kiore do eat tree weta eggs (Ussher 1995), as well as adult weta, so tree weta populations may have been suppressed by kiore and increased quickly following their removal.

In summary, these data support the hypothesis that the diet and feeding ecology of tuatara are significantly altered in the presence of kiore. However, the variable response found between habitats in this study (Table 5) also tentatively supports proposals that the intensity of competitive effects of kiore are linked to specific habitat types. In doing so, these results also support models which propose that the impacts of kiore are linked to the age of forest following disturbance and reduce as forests mature (Craig 1986). As Craig (1986) suggested, competition may only be a feature of regenerating forest.

**Historic impact model**

Plotting density of tuatara, degree of historical habitat modification and presence of kiore shows that tuatara will recover from habitat modification alone but are suppressed if kiore are also present (Fig 4). Craig (1986) first speculated on the significance of interactions between kiore presence and past human-induced habitat modification. He proposed a sliding scale of impact for individual islands over time depending on the intensity of past habitat destruction. This encourages investigation of factors which may modulate kiore impact (tackling causes of decline) rather than merely confirming present effects through monitoring eradications (tackling symptoms of decline). Implicit within the assumption that present impact may not represent past impact is that changes in habitat factors will modify the potential of kiore to impact on wildlife. Craig’s model (hereafter referred to as the Historic Impact Model) can be best represented by considering trends in kiore population dynamics and how these relate to the three levels of hypothesis testing required to establish detrimental kiore impact on native fauna. The potential for impact, intensity of impact and significance of impact need consideration in each of three general successional stages following forest removal - early and mid regenerating forest and mature forest (Fig. 5).
Table 5. Summary of changes in environmental invertebrate abundance and tuatara dietary characteristics following the removal of kiore from two successional stages of forest on Lady Alice Island. See text and previous tables for numerical data.

<table>
<thead>
<tr>
<th>Response</th>
<th>Regenerating forest</th>
<th>Mature forest</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Summer</td>
<td>Autumn</td>
</tr>
<tr>
<td>Increased invertebrate abundance ?</td>
<td>no data</td>
<td>no data</td>
</tr>
<tr>
<td>Increase in number of prey consumed ?</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Increase in size of prey consumed ?</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Increased foraging Success ?</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Dietary Contraction (see text for definition)</td>
<td>Yes (moderate)</td>
<td>Yes (large)</td>
</tr>
<tr>
<td>Prey switching</td>
<td>Low</td>
<td>High</td>
</tr>
</tbody>
</table>

Figure 4. Effects of habitat modification and presence of kiore on the density of tuatara on islands. Islands with no history of kiore presence indicated by closed diamonds, those inhabited by kiore shown by open circles. Degree of habitat modification defined as: 1 (low) - Maori visitation only; 2 - Maori inhabitation and sustained forest clearance; 3 - Maori inhabitation and European infrequent forest clearance; 4 - Maori inhabitation and European farming (and stock) and/or sustained forest clearance; 5 (high) - Maori inhabitation, European farming and sustained forest clearance.
farming and introduced predators or browsers (goats, pigs, cats). Data from Cree & Butler 1993; Robert Bracey pers. comm.).

**Potential for impact**

The annual cycles characterising kiore population dynamics show breeding during late spring, population increase over the summer months to peak during autumn followed by sudden decline and low densities throughout winter (Fig. 5a; Bunn & Craig 1989; Moller & Craig 1987; Roberts & Craig 1990). Cycles are regulated by both the availability of high protein seeds (Craig & Bunn 1989) and the availability of suitable shelter (Roberts & Craig 1990). As habitats regenerate from severe modification such as burning and farming, seed becomes more seasonally dispersed (Fig 5b; Bunn & Craig 1989; Speed 1986) and shelter (in the form of cavities and leaf litter) increases. Comparisons between intensively modified farmland through to mature forest show successively lower kiore densities, dampened population fluctuations and reduced reproductive output as specific habitats mature (Craig 1986; Moller & Craig 1987; Roberts & Craig 1990). Thus, both the average density and maximum density of kiore is greater immediately following habitat modification and reduces as forest matures due to the availability and wider seasonal dispersion of seeds and shelter.

Kiore will preferentially forage on high protein fruits and seeds, but switch increasingly to less favoured vegetation and animal foods as preferred sources become or are scarce as in regenerating forest (Roberts & Craig 1990; Ussher unpubl. data). However, foraging is density-dependent and kiore will exploit abundantly available foods irrespective of either food type (animal, seed) or season. Analysis of kiore diets from Lady Alice I. show that invertebrates dominate the animal component (Newman & McFadden 1990; Ussher unpubl. data - 97% of items), a pattern repeated on other islands and habitats (Bettesworth 1972; Bunn 1979; Bunn & Craig 1989).

Thus, patterns of kiore population dynamics and feeding from this and other studies indicate that potential for impact (in general) on native fauna will be more severe in early regenerating habitats than more mature habitats. Also, potential impact is greater during the autumn months than during winter or spring, as kiore populations reach peak densities over autumn, but increasingly eat animal sources as seeds and fruits become scarce. Because invertebrate diversity and abundance is proportional to habitat maturity (Crisp et al. 1998; Moeed & Meads 1992), and kiore population density and level of fluctuation declines as habitats mature, the potential scale of kiore impact is exacerbated for early, compared to later, regenerating habitats.
Figure 5. Historic Impact Model predicting the effects of kiore on native fauna over time as forest cover regenerates following human-induced disturbance. Basic dynamics of kiore populations (A: sources; Bunn 1979, Craig 1986, Moller & Craig 1987, Roberts & Craig 1990), the distribution of fruits and seeds between habitats (B: sources; Bellingham 1984, Speed 1986, Astrid Dijkgraaf pers. comm.) and their role of regulators of kiore population cycles (C: sources; Craig & Moller 1987, Craig & Bunn 1989, Roberts & Craig 1990) are established data. From these, predictions of impact intensity on native fauna (C, D) and the significance of impact on the persistence of native fauna in the presence of kiore (E), can be made. Habitat types used to construct trends are: early successional - rank grassland; mid successional - open canopy kanuka; late successional mature) - coastal mixed broadleaf. Predictions of persistence of native fauna (E) are generalised for species considered vulnerable to impact by kiore and which naturally have larger population sizes in regenerating habitats than mature habitats. Trends within habitat types for A,B,C,D show predictable annual cycles. Predicted decline of native populations (E) is compounded oversuccessive years or by return of habitat to early successional stage through fire and other disturbance.
**Intensity of impact**

If the release of tuatara from food competition with kiore is regarded as an indicator of indirect impact intensity, the results of this study provide quantitative support for hypotheses generated in the Historic Impact model (Fig. 5c). Intensity of competition is indeed greater in the younger forest and further, impact is more intense during the autumn than the summer months for both habitats. Differences in tuatara feeding response between forest types reflects the degree of kiore impact on resident invertebrate populations.

As kiore competition for invertebrates increases, changes in tuatara feeding behaviour will progress through a number of stages. Low level competition will result in tuatara redistributing predation pressure within a similar range of prey items. Higher levels of competition will see tuatara denied access to favoured prey, and they will therefore switch increasingly to new prey groups. Further competition will see a reduction in the size and number of items accessible for consumption. Under intense competition, the ability of tuatara to catch prey will be compromised and invertebrate populations will be markedly suppressed by kiore predation. Kiore may also influence the ability of tuatara to catch prey by triggering anti-predator behaviour in adult tuatara (see above). If so, foraging success may be compromised even when invertebrate abundance is high and other foraging characteristics do not change. At each level of competition, the switch by kiore to predominantly invertebrate foods during autumn will accentuate trends seen during summer.

Tuatara in regenerating forest experienced intense levels of competition which lowered both quantity (number) and quality (size and taxa) of food consumed (Table 5). By comparison, the intensity of competition was considerably lower for tuatara in mature forest where neither food quality nor quantity was influenced at levels approaching patterns found in regenerating forest. Tuatara in both habitats were prevented from feeding by the presence of kiore. The minimal impact of kiore on other aspects of tuatara foraging in regenerating forest suggests that kiore do not prevent feeding through food competition but act through some other mechanism, such as eliciting anti-predator behaviour. The greater level of food denial experienced by tuatara in regenerating forest could be attributed to either intrinsically lower levels of invertebrates or greater densities of kiore either interfering with foraging or competing for food at levels approaching total exclusion, or an interaction of these two factors. Thus, kiore competitively exclude tuatara from food in the regenerating forest but jointly exploit invertebrates with little measurable effect on tuatara feeding efficiency in the mature forest. In both habitats, kiore directly interfere with tuatara foraging success, perhaps by disrupting hunting activity.
The influence of kiore is also clearly more intense during autumn compared with summer (Table 5). The higher degree of prey switching by tuatara during autumn in both habitats following kiore removal may be due to the peak densities of kiore over this season and the switch in kiore diet from predominantly plant-derived foods to invertebrate alternatives (see above). Changes in specific tuatara food groups are most marked for tree weta. Their sudden appearance in autumn diets following kiore removal reinforces the seasonal distinction in the intensity of impact by kiore.

**Significance of impact**

An important constraint in this study is the lack of detailed knowledge of tuatara density in each study site. While efforts were made to sample areas of comparative density of tuatara between regenerating and mature forest, differences in local density, and hence intra-specific competition for food, may have contributed to the dietary responses seen between sites. However, the differences in dietary response still encourage speculation on the relative impacts of kiore on tuatara and fauna, especially in the context of management options for restoring native populations. Further study will be necessary before comment on competition by kiore can be generalised to other habitats and locations with confidence.

Also, this study assessed dietary competition only for adult tuatara. Juvenile tuatara on other islands in the absence of kiore are known to eat considerably smaller prey (from 10mm to below 3mm; Fraser 1993) than adults. Kiore will also eat small prey, especially when available in large quantities (Ussher unpubl. data), so it is probable that food competition between kiore and juvenile tuatara exists and may indirectly influence juvenile survivorship and recruitment into populations.

Seasonal variation in the intensity of food competition has important implications for breeding success of tuatara and the persistence of other fauna. Tuatara mate during late summer/early autumn and females deposit shell around the eggs in utero over winter (Cree et al. 1996). Food quality may determine reproductive frequency for tuatara (Cartland-Shaw et al. 1998) and therefore, may also influence the quality of eggs produced. Kiore in this study appeared to successfully out-compete tuatara for high quality (large source of protein) foods such as tree weta during autumn and this may indirectly influence reproductive success by, for example, interfering with vitellogenesis or inhibiting the shelling process.

The distinct differences in impact intensity noted between habitats also has consequences for past and for present impact of kiore on native populations. Adult tuatara in regenerating habitats face greater competition from kiore for food (this study) than tuatara in more mature habitats. From the
results of this study, and assuming that competition reduces tuatara condition and breeding frequency, it would be expected that once kiore are removed increases in tuatara body weight and clutch characteristics (size, frequency) would be significantly greater for animals in regenerating habitats compared to mature habitats. Newman et al. (1994) attribute reduced clutch size observed for tuatara on kiore-inhabited Lady Alice I. to depletion of food resources. However, that study grouped tuatara from both regenerating and mature forests, masking any difference in clutch size between habitats.

The results from this study also generate working hypotheses for the dynamics of predation by kiore on native fauna. Intensity of predation by kiore on tuatara eggs and young should be greater in regenerating habitats because of greater fluctuations in kiore population density and increased competition for animal food sources (both vertebrate and invertebrate), especially during autumn when preferred plant foods become scarce (Fig 5d). In contrast, direct impacts of kiore on tuatara should be less in mature habitats and may be at levels where tuatara populations are suppressed at low densities instead of in continual decline (Fig 5e; c.f. Cree & Butler 1993). Studies which directly measure the breeding success of tuatara need to be conducted. With current technology (radio transmitters and miniature cameras) studies should be able to provide valuable information on the hatching success of eggs and recruitment by juveniles into populations in different habitats in the presence and absence of kiore.

The relict status of tuatara populations seen on some other kiore-inhabited islands (Cree & Butler 1993, Cree et al. 1995) may be the result of intense and frequent habitat modification known to have occurred (Cree & Butler 1993) with subsequent direct effects on tuatara population levels and indirect effects from repetitive cycles of elevated kiore impact. Indeed, past island history is often considered in this context (e.g. Newman & McFadden 1990) although ultimate predictions of tuatara status on less modified islands in the presence of kiore remain centred on decline and eventual extinction of tuatara (e.g. Cree et al. 1995).

**Implications for management**

'An association between a disappearance or reduction of a local population and a possible cause is not in itself a diagnosis. It is a speculation leading to hypotheses to be confirmed by testing.'

Caughley & Gunn 1996

Debate on kiore impact on fauna is dominated by preconceived assumptions and a lack of detailed study aimed at the level of hypothesis testing strongly advocated by Caughley and Gunn (1996). The Tuatara Recovery Plan (Cree & Butler 1993) appears to be a clear example where untested
theories form the basis of management strategies without acknowledging the reliability of information. Subsequent management actions have seen kiore eradicated (by the time this study is completed) from 6 of the 9 tuatara islands once supporting kiore, with 2 more islands planned for eradication in the near future (DoC 1994).

**Approaches to species recovery planning**

Two general management approaches can be proposed for the implementation of species recovery (c.f. the science of species recovery which includes single species or ecosystem approaches). In 'Biologically Dominated Management', the biology required for recovery dominates management actions and focuses research strategy on defining factors which currently contribute to population suppression or decline (Fig. 6a). The objectivity associated with understanding *how* and *when* populations have arrived at their current status is instead replaced with identifying *why* populations differ and finding immediate and usually single solutions to ensure population enhancement. Biologically centred management, such as this, identifies the species as the only recipient of conservation action and does not consider alternative strategies which might also lead to long term sustainable management of wildlife (Hartley 1997). A second approach ('Integrated Co-management') integrates social and economic philosophies, as well as biological, into recovery options and relates specific actions to a wider framework of sustainable species management (Fig. 6b).

The current structure of New Zealand species management systems appears to perpetuate pragmatic actions centred on the recovery of single species. Individual Species Recovery Teams are driven by the guiding principle to 'maintain and enhance' distributions and to identify actions necessary to enable species recovery (DoC 1998a), thus predisposing management more towards principles of Biologically Dominated Management than those of Integrated Co-management. Tuatara management is neither completely pragmatic nor bio-centric as public access to some islands has recently been included as a priority (Cree and Butler 1993) and tuatara have recently been re-introduced to Somes/Matiu I. situated in the centre of the capital city's harbour (DoC 1998b). Research focus for investigating kiore interactions with tuatara remains largely biological. Initial calls for the investigation of the nature of kiore interactions with tuatara (Newman 1983; Newman & McFadden 1990) appear to have been superseded by eradication recommendations in the subsequent national tuatara strategy - the Tuatara Recovery Plan (Cree & Butler 1993). The adoption of eradication of kiore as the only management approach was also probably helped by advances in eradication methodology which allowed kiore to be removed from much larger islands than before, and the perceived benefits of eradication for many elements of the ecosystem - not just tuatara. However, the Recovery Plan sets a peculiar standard by which to conduct wildlife research.
It identifies research into kiore interactions with tuatara as a priority but only under the auspices of kiore eradication from all tuatara islands (i.e. *post hoc* justification).

Some authors who support this approach cite the need for 'precautionary management' (Cree *et al.* 1995), where the body of circumstantial evidence linking kiore to tuatara decline, and the risk of tuatara population loss provides extenuating circumstances demanding eradication. While the application of 'precautionary management' may be laudable for some tuatara populations, the
endorsement of kiore removal on all islands where tuatara and kiore co-exist and as the only management tool has consequently narrowed hypothesis testing to focus on confirmatory studies rather than exploratory experiments (e.g. Newman et al. 1994; Cree et al. 1995). In so doing, management actions have relegated the role of science from driving the management of remaining co-existent populations to merely recording interactions for historical reference.

A posteriori hypothesis testing will probably demonstrate that, over time, the removal of kiore results in increased tuatara recruitment and abundance. However, this approach is unable to demonstrate whether the removal of kiore is the only answer to enhancing tuatara populations on kiore-inhabited islands. This is especially unsatisfactory when the intensity of kiore impact on adult tuatara appears habitat-dependent (this study) and thus over time as forest communities mature, tuatara could be expected to increase anyway. Prior to management intervention, most (6 out of 9) tuatara populations with kiore present comprised only a few (low 10s) aged individuals and extinction appeared imminent. Captive breeding programmes now offer realistic solutions to quickly increasing population size. Removal of kiore is seen as a prerequisite to re-introducing founders and their offspring, but captive breeding also offers opportunities to test the need for kiore eradication through planned experiments.

An option considered in the draft Tuatara Recovery Plan (Cree 1990) for Little Barrier Island (3082 ha), before the advent of improved rodent eradication techniques for larger islands, was the captive rearing and release of tuatara at sizes large enough to be beyond the size range of prey taken by kiore in the hope of establishing a self-sustaining population. There is no reason why such a programme cannot be implemented now for tuatara or expanded to trial the (re)-introduction of tuatara, lizards and invertebrates on islands which still support kiore. Indeed, at a recent New Zealand herpetological conference there was support for implementing such programmes to directly test the value of kiore-inhabited islands as viable restoration sites for native reptiles. This may be the only way that further study of kiore interactions with tuatara can be achieved, as kiore have now been removed from all islands supporting large populations of tuatara. The three remaining islands with sympatric populations of tuatara and kiore support only relict populations of tuatara and are unlikely to be suitable for studying inter-specific interactions or as experimental controls for studies on islands from which kiore have recently been removed (c.f. Cree et al. 1995). However, the draft revision of the current Tuatara Recovery Plan stipulates that the removal of kiore from all remaining islands except Mauitaha (from which only a single mature tuatara has been recorded) is a primary objective for securing existing populations (DoC 1999).
Maori participation in wildlife management

The need to better understand the dynamics of kiore impact on tuatara and other wildlife is becoming more important as traditional Western-centred conservation ethics change to incorporate indigenous peoples' involvement in conservation management (IUCN 1997; Posey 1996). Under Section 4 of the Conservation Act 1987, the DoC must 'interpret and administer the Act so as to give effect to the principles of the Treaty of Waitangi'. Embedded within these principles is the protection of possessions valued by Maori. It is not unreasonable to suggest that the kiore, as an integral component of Maori cultural history and genealogy (see introduction), should be managed through joint co-operation between conservation and Maori interests.

The inability of the DoC to effect co-management for kiore is well documented (Ngatiwai Trust Board 1995; Roberts 1995), as is the frustration of Maori at being excluded from decision-making processes (Taiepa et al. 1997) or being denied informed choices on the future of kiore-inhabited islands (e.g. Ngatiwai Trust Board 1995). Moreover, the adoption of unsupported evidence by conservation authorities and promotion of indirectly inferred impacts when seeking approval of Maori stakeholders can legitimately be seen as coercion (Hartley 1997; Taiepa et al. 1997). However, the recent co-management initiatives proposed by DoC (e.g. DoC 1998c; Ngai Tahu 1997) and the greater desire of the DoC to involve Maori in management decisions at the local level are encouraging signs that previous philosophies on wildlife co-management are changing. Indeed, legislative backing and increasing calls for co-management regimes in New Zealand (e.g. Moller 1996; Taiepa et al. 1997) suggest that decisions which incorporate the values of all stakeholders into wildlife management decisions (be it for native or introduced species) will be the way of the future (see also Brussard et al. 1994).

The importance of kiore to Maori differs on a regional basis. Some iwi actively support the removal of kiore from islands within their tribal boundaries for the purpose or opportunity of restoring tuatara (and other wildlife) populations (P. Gaze pers. comm.). The Draft Kiore Management Plan (DoC 1994) provides options for transferring kiore from islands facing eradication to low biological value islands should iwi wish to retain distinctive kiore populations. All remaining tuatara populations in the presence of kiore are on northern New Zealand offshore islands in the regional jurisdiction of one iwi. Issues of concern for that iwi (Ngatiwai) about the management of kiore on tuatara islands include the eradication methodology used (poisons), effective involvement in decision-making processes for natural resources, revival of traditional kiore harvesting practices, and interest in investigating alternative methods to kiore eradication to achieve restoration of native wildlife.
Indeed, these issues suggest a further important variable for the Historic Impact Model. Removal of Maori customary harvesting would have occurred immediately prior to habitat destruction by European settlers. This would have further exacerbated the negative effects of kiore.

Conclusions

Overall, management of tuatara on islands with sympatric populations of kiore has been overwhelmingly performed in an environment where scientific objectivity is scarce and the framework purely biological. In today’s conservation environment, tuatara management demands broader considerations including, in some places, that of species in addition to tuatara (such as kiore), and for integrating philosophical approaches to conservation other than biological considerations (e.g. social, cultural, economic). The results of this study should therefore have direct implications for the future management of the remaining islands on which tuatara and kiore co-exist and indeed, for broader considerations of kiore impact in native ecosystems.

The suggestion that eradication of kiore from islands may not be the only management tool necessary to restore native fauna offers an opportunity for closer co-operation between conservation managers and Maori stakeholders in wildlife. Further, it encourages the investigation of the viability of managing kiore populations sympatric with tuatara for the purpose of sustainability instead of eradication. Testing well formed hypotheses and applying these to management in the absence of speculation will be vital to realising both of these goals.

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APPENDIX 2.1:
Food groups which showed no significant change in frequency of occurrence (% presence/absence in tuatara stomachs) following kiore removal from Lady Alice I. mature forest. Statistics in bold used to infer significance of change. SIG = significant change; NS = not significant.

<table>
<thead>
<tr>
<th>Food Group</th>
<th>Habitat (succession)</th>
<th>Effect</th>
<th>df</th>
<th>Chi-Square</th>
<th>Probability</th>
</tr>
</thead>
<tbody>
<tr>
<td>Worms (Class Oligochaeta)</td>
<td>mid</td>
<td>season</td>
<td>1</td>
<td>73.91</td>
<td>&lt;0.001 SIG</td>
</tr>
<tr>
<td></td>
<td>late</td>
<td>season</td>
<td>1</td>
<td>0.07</td>
<td>0.80 NS</td>
</tr>
<tr>
<td></td>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>0.03</td>
<td>0.85 NS</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>3.19</td>
<td>0.07 NS</td>
</tr>
<tr>
<td>Slugs/Snails (Class Gastropoda)</td>
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<td>season</td>
<td>1</td>
<td>165.32</td>
<td>&lt;0.001 SIG</td>
</tr>
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</tr>
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<td>0.01</td>
<td>0.93 NS</td>
</tr>
<tr>
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<td>1</td>
<td>0.14</td>
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<td></td>
<td>late</td>
<td>season</td>
<td>1</td>
<td>0.10</td>
<td>0.75 NS</td>
</tr>
<tr>
<td></td>
<td></td>
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<td>1</td>
<td>0.21</td>
<td>0.65 NS</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.21</td>
<td>0.65 NS</td>
</tr>
<tr>
<td>Slaters (O. Isopoda)</td>
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<td>0.18</td>
<td>0.67 NS</td>
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<td>0.25</td>
<td>0.62 NS</td>
</tr>
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<td>0.01</td>
<td>0.91 NS</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.01</td>
<td>0.91 NS</td>
</tr>
<tr>
<td>Amphipods (O. Amphipoda)</td>
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</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.99</td>
<td>0.32 NS</td>
</tr>
<tr>
<td>Bristletails (O. Microcoryphia)</td>
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<td>season</td>
<td>1</td>
<td>0.72</td>
<td>0.40 NS</td>
</tr>
<tr>
<td></td>
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<td>season</td>
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<td>0.65 NS</td>
</tr>
<tr>
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<td></td>
<td>kiore removal</td>
<td>1</td>
<td>0.21</td>
<td>0.65 NS</td>
</tr>
<tr>
<td>Insect Larvae</td>
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<td>season</td>
<td>1</td>
<td>68.70</td>
<td>0.00 SIG</td>
</tr>
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<td></td>
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<td>season</td>
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<td>0.07</td>
<td>0.80 NS</td>
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<tr>
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<td>kiore removal</td>
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<td>1.59</td>
<td>0.21 NS</td>
</tr>
<tr>
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<td></td>
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<td>0.20 NS</td>
</tr>
<tr>
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<td>Season</td>
<td>Occurrence</td>
<td>Control</td>
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</tr>
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<td>--------</td>
<td>---------</td>
<td>------------</td>
<td>---------</td>
<td>---------------</td>
</tr>
<tr>
<td><strong>Beetles</strong> (O. Coleoptera)</td>
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<td>0.80</td>
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</tr>
<tr>
<td></td>
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<td>season</td>
<td>1</td>
<td>1.22</td>
<td>0.27 NS</td>
</tr>
<tr>
<td><strong>Earwigs</strong> (O. Dermaptera)</td>
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<td>season</td>
<td>1</td>
<td>0.80</td>
<td>0.37 NS</td>
</tr>
<tr>
<td></td>
<td>late</td>
<td>season</td>
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<td>1.22</td>
<td>0.27 NS</td>
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<td>0.14</td>
<td>0.70 NS</td>
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<td>0.10</td>
<td>0.75 NS</td>
</tr>
<tr>
<td><strong>Stick Insects</strong> (O. Phasmida)</td>
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<td>season</td>
<td>1</td>
<td>0.00</td>
<td>1.00 NS</td>
</tr>
<tr>
<td></td>
<td>late</td>
<td>season</td>
<td>1</td>
<td>0.38</td>
<td>0.54 NS</td>
</tr>
<tr>
<td></td>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>0.21</td>
<td>0.65 NS</td>
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<td></td>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>0.28</td>
<td>0.60 NS</td>
</tr>
<tr>
<td><strong>Mantids</strong> (O. Mantodea)</td>
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<td>season</td>
<td>1</td>
<td>0.18</td>
<td>0.67 NS</td>
</tr>
<tr>
<td></td>
<td>late</td>
<td>season</td>
<td>1</td>
<td>0.25</td>
<td>0.62 NS</td>
</tr>
<tr>
<td></td>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>not eaten</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>not eaten</td>
<td>-</td>
</tr>
<tr>
<td><strong>Flies</strong> (O. Diptera)</td>
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<td>season</td>
<td>1</td>
<td>0.00</td>
<td>1.00 NS</td>
</tr>
<tr>
<td></td>
<td>late</td>
<td>season</td>
<td>1</td>
<td>0.39</td>
<td>0.53 NS</td>
</tr>
<tr>
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<td></td>
<td>kiore removal</td>
<td>1</td>
<td>0.11</td>
<td>0.74 NS</td>
</tr>
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<td>kiore removal</td>
<td>1</td>
<td>0.11</td>
<td>0.74 NS</td>
</tr>
<tr>
<td><strong>Cicadas</strong> (O. Homoptera)</td>
<td>mid</td>
<td>season</td>
<td>1</td>
<td>2.01</td>
<td>0.16 NS</td>
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<tr>
<td></td>
<td>late</td>
<td>season</td>
<td>1</td>
<td>1.46</td>
<td>0.23 NS</td>
</tr>
<tr>
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<td></td>
<td>kiore removal</td>
<td>1</td>
<td>0.61</td>
<td>0.44 NS</td>
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<td>1</td>
<td>0.05</td>
<td>0.82 NS</td>
</tr>
<tr>
<td><strong>Plant Suckers</strong> (O. Hemiptera)</td>
<td>mid</td>
<td>season</td>
<td>1</td>
<td>0.14</td>
<td>0.70 NS</td>
</tr>
<tr>
<td></td>
<td>late</td>
<td>season</td>
<td>1</td>
<td>0.10</td>
<td>0.75 NS</td>
</tr>
<tr>
<td></td>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>0.21</td>
<td>0.65 NS</td>
</tr>
<tr>
<td></td>
<td></td>
<td>kiore removal</td>
<td>1</td>
<td>0.28</td>
<td>0.59 NS</td>
</tr>
</tbody>
</table>
Plate 3
Moutohora, showing location of study sites and physical characteristics of vegetation and burrow factors tested. Artificial burrows were installed in each study site.
Abstract

Well planned experiments are critical for identifying which factors determine the success or failure of re-introductions. Testing habitat needs of species offers increased effectiveness for subsequent re-introductions and species management in general by identifying alternative management strategies and refining site selection criteria for future re-introductions.

Habitat needs, thought to be important for the successful re-introduction of tuatara, a reptile endemic to New Zealand, were tested in an experimental re-introduction to Moutohora (Whale Island). Survivorship, body condition, and dispersal of founders were monitored for 16 months after release. Re-introduction success was tested in habitats which differed by vegetation age (early and mid-regenerating forest) and dispersion of natural underground refuges (evenly spaced seabird burrows at high densities and clumped burrows at high densities).

Vegetation age and burrow dispersion had no effect on survivorship or condition of tuatara, but burrow dispersion did influence animal movement. Tuatara moved significantly greater distances from release locations in areas where burrows were clumped and dispersed further from their release points throughout the study. By comparison, tuatara in evenly distributed burrow sites were more likely to remain within release areas. Despite statistical differences in movement of tuatara between sites, all release groups were considered to be reproductively viable after 16 months following release. Artificial seabird burrows, trialed in each release site, were used by some animals for up to 16 months, but were deserted by most within 60 days after release.
Experimenting with release environments for tuatara could enable restoration to mainland sites. Full exploitation of the opportunities presented by identifying habitat plasticity (and constraints) will require the refinement of current site selection and re-introduction assessment criteria. Criteria which allow re-introduction for principally non-biodiversity goals - such as social, educational or cultural - and consider multiple re-introductions with diverse goals will better achieve species conservation than conservative approaches which select sites based only on existing habitat characteristics.

Key words: habitat requirements, re-introduction, tuatara, Sphenodon, Moutohora, Whale I.

INTRODUCTION

Re-introduction or translocation of species is increasing in popularity as wildlife managers seek to conserve threatened species *in situ* and restore communities and ecosystems (Chapter 1, Griffith *et al.* 1989). Successful population establishment benefits from correctly identifying suitable habitat for founders (Griffith *et al.* 1989). Often, however, few data exist other than habitat associations of remnant populations to guide the selection of new sites (c.f. Gray & Craig 1991). Even under such conservative selection procedures, the exact factors which contribute to the success or failure of re-introductions can only be known with certainty (and appropriate management action for future transfers undertaken), if such re-introductions are planned as experiments (Armstrong *et al.* 1994, Armstrong & McLean 1995). Undertaking re-introductions to new habitats as trials (where scientific replication and design are not considered) may be more cost effective in the short term, but will result in increased long-term costs and reduced options for future management (Chapter 1). Responsible management demands increased effectiveness in decision-making processes (Craig & Stewart 1994). Recent conservation efforts for the tuatara *Sphenodon* spp., a medium-sized, nocturnal reptile endemic to New Zealand, illustrate how experimental re-introductions can help refine criteria for site selection and thus increase the efficiency and effectiveness of threatened species management.

Human settlement and exotic mammalian predators have reduced tuatara from a widespread mainland distribution to only 32 offshore islands (Cree & Butler 1993, Cree *et al.* 1995a). The national management strategy for tuatara, the Tuatara Recovery Plan (Cree & Butler 1993), identifies re-introduction to predator-free islands, and more recently to mainland sites (DoC 1999a), as priority measures for tuatara restoration. Only general recommendations for criteria important for selecting re-introduction sites are included. Suggested criteria such as the presence of seabird colonies and open canopy forest reflect habitat of existing remnant tuatara populations. Extensive physiological and ecological research has established several biological attributes of tuatara, such
as preference for existing underground refuges, vulnerability to desiccation and density dependant foraging (see below), which may increase their probability of successful establishment in some habitats than others. Testing the importance of these perceived habitat needs will help determine the habitat plasticity of tuatara and may considerably increase the diversity of habitats and locations where tuatara can be restored.

The success of a tuatara re-introduction may be influenced by the availability of refuges. Remnant tuatara populations are closely associated with seabird colonies and tuatara use these burrows as day-time retreats. Wild tuatara will dig their own burrows but captive animals preferentially seek out existing refuges before constructing their own (if at all) over a period of days or weeks (M. Bell & C. Smuts-Kennedy pers. comm.). Both genders are site attached and maintain home ranges, which overlap, of about 15-20m from core burrows (Newman & McFadden *in press*, pers. obs.). Little is known about how mates find each other, but courtship appears to be conducted using only visual stimuli and no vocalisations are used (Cree & Thompson 1988). Together, these traits suggest that any factor, such as the dispersion of refuges, which causes founders to disperse widely upon re-introduction may compromise population viability by reproductively isolating individuals. Therefore, tuatara released into areas where burrow refuges are clumped can be expected to disperse further from release sites than those released into areas where burrows are evenly dispersed.

Air temperatures within habitats may also influence the chances of successful population establishment. Tuatara are vulnerable to excessive cutaneous water loss (L. Hill pers. comm. in Barwick 1982) and therefore, in habitats which are prone to severe drying, micro-climatic environment may not favour survival or breeding. Micro-environmental temperatures determine incubation rates of tuatara eggs (Thompson 1990) and gender of tuatara (Cree *et al.* 1995b). Tuatara in the wild seek open areas in which to nest (Cree & Thompson 1988) and hatching success of eggs in open areas is greater than in forest habitats where soil temperatures are cooler (Cree *et al.* 1989, Thompson *et al.* 1996). Together, these traits suggest that tuatara re-introduced to closed forest which is vulnerable to desiccation, such as early-regenerating forest, may increase the risk of founder mortality and reduce breeding success.

Similarly, tuatara feeding ecology suggests that founder populations will be more likely to establish under open canopy forest (such has secondary mid-successional regenerating forest) than closed canopy forest (such as early-successional regenerating forest). Early-regenerating forest of manuka (*Leptospermum scoparium*) or kanuka (*L. ericoides*) are generally dense monotypic stands with poorly developed leaf litter, low ground light levels and high evapo-transpiration rates leading to dry soils and leaf litter. Such habitats support lower beetle diversity than more developed forest types (Crisp *et al.* 1998), and may support a lower diversity and abundance of tuatara food items than
more mature forest (C. Green unpubl. data). Tuatara employ a ‘sit and wait’ hunting strategy (Walls 1981) and prey capture is dependent on environmental density of prey (Ussher 1999). Therefore, habitats which support fewer prey items (diversity and/or abundance) may present a greater risk of founder mortality or reduced breeding condition of individuals, or may result in relatively greater dispersal rates of founders. Tuatara released into habitats where food availability is below a critical threshold should disperse further than those in habitats where food is in excess of requirements. Also, tuatara released into early-regenerating habitats will have a greater probability of mortality or loss of condition than those released into more mature habitats.

Recent sub-fossil evidence from mainland sites showing high numbers of tuatara in the absence of seabirds (Worthy 1994) suggests that the association between seabird colonies and tuatara densities may be an artefact of island refuges. Additionally, the importance of vegetation type and the availability of retreats in determining outcomes of re-introductions of tuatara is seen as a research priority (Cree & Butler 1993). The recent re-introduction of tuatara to Moutohora (Whale Island) in the Bay of Plenty, New Zealand presents the first opportunity to experimentally test the relative importance of these two habitat factors in determining the short-term success of re-introduction programmes for tuatara.

RE-INTRODUCTION AND SITE DESCRIPTION

Source island

Moutoki Island lies 7km to the west of Moutohora (Plate 1) and is approximately 0.8ha, of which 0.5ha is habitable by tuatara. The island comprises an eroded volcanic plug approximately 15m in height from which steeply sloping cliffs or earthen banks descend to the foreshore (Plate 1). The eastern end of the island comprises a flat area of ground (0.4ha) of boulder banks, a developed soil structure and established Hymenanthera novae-zealandiae and Taupata coprosma repens forest up to 3.5m in height. The eastern flats are heavily pitted with seabird burrows (principally grey-faced petrel Pterodroma macroptera gouldi and blue penguin Eudyptula minor), rock crevices and tuatara burrows (total refuges at approximately 0.5 refuges/m²), and ground cover is sparse.

Founders

Twenty female and 12 male tuatara were caught from either the eastern flats of Moutoki or from the slopes immediately adjacent to the flats over three nights from October 15th-17th 1996. Tuatara were caught at night when active above ground. Only mature adult tuatara were selected for transfer to Moutohora. Adult male tuatara were defined as being between 190mm and 220mm snout vent length (SVL) and adult females between 165 and 190 mm SVL. Also, only non-gravid female tuatara
(determined by gently palpating the abdomen to check for eggs) were transferred because gravid tuatara lay eggs during late spring (October -November) and can travel up to 200m in search of suitable nest sites (Daugherty & Cree 1990).

Release island

Moutohora is a steep 143ha dormant volcano situated 9.5km off the north-west coast of Whakatane, Bay of Plenty. Maori and European occupation of the island for living, farming and resource exploitation, as well as the introduction of goats, rabbits and Norway rats Rattus norvegicus, have severely modified the forest cover over the last 300 or more years. Elders of the local Maori tribe, Ngati Awa, recall handling tuatara on the island in the early 1900’s, but no sign has been seen since and the population is assumed to be extinct.

Goats were eradicated in 1977 and rats and rabbits by 1987/1998 (DoC 1999b) and forest has since regenerated. Vegetation on the island now comprises a mix of rank grasslands, early secondary regenerating kanuka forest and mid-regenerating kanuka/ mixed broadleaf forest (including species such as kohekohe Dysoxylum spectabile, mahoe Melicytus ramiflorus, cabbage tree Cordyline australis, kawakawa Macropiper excelsum and puriri Vitex lucens). Remnant populations of burrowing Grey-faced Petrel, which once inhabited the island in the hundreds of thousands (van der Wouden 1994), are now expanding into regenerating areas.

Release sites

Founder tuatara were released on Moutohora over a 1 hour period during the evening (close to the primary natural activity period of tuatara) of October 18th 1996. The release was undertaken in mid-spring (October) when invertebrate foods for tuatara are abundant (C. Green pers. comm., pers. obs.), and rainfall still frequent (Table 4). Release before summer was considered important as rainfall is low and this may exacerbate desiccation of tuatara and hence reduce survivorship.

Tuatara were monitored for 16 months following release (until March 1998), with monitoring visits of 1-2 weeks duration undertaken once every season.

Tuatara were released into four sites on Moutohora, determined by the distribution of seabird burrows and the degree of development of forest cover. Two release sites were located in early-regenerating forest (Table 1), one supporting evenly located seabird burrows at approximately 0.5 burrows/ m² (Table 1; even burrow site) and one where seabird burrows were clumped at high densities (0.5 burrows/ m²) within a landscape of low burrow density (0.06 burrows/ m²; clumped burrow site). Similarly, of two release sites in mid-regenerating forest (Table 1), one supported evenly distributed burrows and the other clumped burrows.
Table 1. Seabird burrow, vegetation characteristics and numbers of tuatara released (male and female) in four sites on Moutohora. Data from Ussher (unpublished) vegetation analysis of study sites.

<table>
<thead>
<tr>
<th>Site</th>
<th>Burrow density (no./m²) (mean +/- S. E.)</th>
<th>Principle tree spp. throughout site</th>
<th>Density of stems (no./m²) (mean +/- S. E.)</th>
<th>Number of tuatara released</th>
</tr>
</thead>
<tbody>
<tr>
<td>Early, even</td>
<td>0.47 ± 0.17 (throughout)</td>
<td>Kanuka (<em>Kunzea ericoides</em>)</td>
<td>9 ± 1.2</td>
<td>8 (3M +5F)</td>
</tr>
<tr>
<td>Early, clumped</td>
<td>0.46 ± 0.08 (clumps)</td>
<td>Kanuka</td>
<td>9 ± 3.3</td>
<td>8 (3M +5F)</td>
</tr>
<tr>
<td></td>
<td>0.06 ± 0.02 ('bare' ground)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mid, even</td>
<td>0.71 ± 0.11 (throughout)</td>
<td>Kanuka and Mahoe (<em>Melicytus ramiflorus</em>)</td>
<td>4.5 ± 0.3</td>
<td>8 (3M +5F)</td>
</tr>
<tr>
<td>Mid, clumped</td>
<td>0.45 ± 0.07 (clumps)</td>
<td>Mahoe</td>
<td>2.5 ± 0.3</td>
<td>8 (3M +5F)</td>
</tr>
<tr>
<td></td>
<td>0.01 ± 0.01 ('bare' ground)</td>
<td>Pohutukawa (<em>Metrosideros excelsa</em>)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Because mixing of release sub-populations during the study could influence individual dispersal and condition, adjacent release sites were separated by a minimum of 500m.

**Release format**

At each site 8 tuatara (5 females and 3 males) were released (Table 1). Each of these sub-populations comprised animals from:

1. each of the three nights of capture on Moutoki - so that possible effects due to duration in captivity were represented at each site;
2. representatives from the extremes of each size class for genders - so that possible post-release responses due to animal size were represented at each site, and
3. individuals caught at 'coastal' and 'inland' locations on Moutoki - so that possible effects of proximity to forest margin/ beach areas were represented at each site.

In each sub-population, two male and three female tuatara were fitted with small back-pack mounted radio telemeters (transmitters). Total weight of the transmitter package was 5g which was at most 1.7% of male or 3% of female body weight. Trials on captive animals indicated no noticeable effects
of transmitter packages on captive tuatara behaviour, feeding or movement and comparison of the above attributes for re-introduced tuatara also indicated no consistent differences between tagged and un-tagged animals (Appendix 3).

Each group of tuatara was released at approximately 10-15m intervals in the geographic centre of release habitats. Within the core areas, five artificial seabird burrows were installed to assess their frequency and duration of use by tuatara. Burrows consisted of a 1.5m length of 115mm diameter plastic drainage tubing (the burrow 'tunnel') buried horizontally in the ground (Plate 3). One end protruded from the ground to act as the burrow hole, the other was fitted to an upturned 10L plastic pail (the burrow 'chamber') from which the bottom had been removed and the resulting cylinder buried vertically to ground level (Plate 3). Burrows were installed 10m apart in two rows at the periphery of the core release area (Figs 3a-d). Tuatara in even burrow sites were released into artificial burrows (5 individuals) or natural seabird burrows (3 individuals), while those in clumped burrow sites were released into artificial burrows (5 individuals) or onto open ground (3 individuals).

METHODS

General
Tuatara have an extremely slow reproductive cycle whereby females lay once every 3-5 years (Newman & Watson 1985, Cree et al. 1991), eggs take up to 12-15 months to hatch (Newman et al. 1979) and juveniles take 9-13 years to attain sexual maturity (Castanet et al. 1988). Therefore, this study tested only short-term responses to founder population viability such as founder survivorship, body condition and degree of dispersal rather than long-term responses such as recruitment and population growth.

Triangulation of tuatara fitted with radio transmitters ('radio-tagged tuatara') generated dispersal information and enabled capture of known individuals to assess body condition. Individual animals were identified by comparison of body length, characteristic natural body markings and natural damage to toes with records of these data taken from individuals prior to release. Toe-clipping, the removal of the terminal joint from toe(s) from each foot to provide a unique long-term marking identification code for each animal, was not used in this study.

Survivorship
Tuatara were considered to have survived the translocation if they were re-captured on or after 150 days following release. This 'critical period' of 150 days was chosen because mortality originating
from summer climatic conditions or the availability of food or shelter in release sites should have been expressed by founders in that period.

**Condition**

Tuatara were caught by hand when either active above ground (at night) or inside known underground refuges (those wearing transmitters). Each individual was weighed, and measurements of SVL, original tail and regenerating tail length recorded.

Changes in animal condition over time were calculated using a condition index which gave a ratio of animal weight to SVL corrected for growth over time:

\[
C_i = \left( \frac{W_i}{L_i} \right) - \left( \frac{W_R}{L_R} \right)
\]

where:
- \(C_i\) = Condition Index
- \(W_i\) = Weight at time \(i\)
- \(L_i\) = SVL at time \(i\)
- \(W_R\) = Weight at release (18 Oct 1996)
- \(L_R\) = SVL at release (18 Oct 1996)

This index assumes that gross morphology of tuatara (e.g. retention of tail) remains constant throughout the study. Tuatara used in this study all had tails which approximated natural length and dimensions and none lost tails during the study.

**Animal movements**

Radio-tagged tuatara were traced to day refuges every second day during monitoring visits to estimate rates of dispersal and degree of site attachment. Minimum distance moved at ground level was measured using a transect tape to the nearest 0.5m. Size of search areas for un-sampled tuatara were estimated by movements of tuatara with transmitters, or those without transmitters whose locations were known. For all release sites a conservative (larger) size for search areas from these estimates was 100m diameter greater than and originating from, the core release area.

Individual tuatara dispersal was calculated as the median distance from release location for each visit. Site attachment of animals was divided into long-term (inter-visit) site fidelity and intra (within-trip) site fidelity. For long-term site fidelity, the first location of each animal and the distance moved
from the last known location (from previous visit) was noted. Short-term site fidelity was calculated by comparing maximum range lengths attained by animals over the course of a sampling visit. Range length immediately following release was not included in analyses because founders may not have immediately adjusted to release environments. Therefore, the first sampling visit included in analyses was 2 months after release.

The proximity of tuatara to potential mates was mapped for the 1997 breeding season (March 1997; autumn). Likely maximum ranges of 15m radius from daytime refuges (based on tuatara territory size in established populations) were plotted for all locations of animals over autumn 1997 as well as the last known record from the previous visit (December 1996) and the first from the following visit (June 1997). These latter locations were plotted because the duration of the breeding season for Northern tuatara is unknown, proximity between individuals prior to the breeding season may be important for initiating subsequent courtship, and animal movement in subsequent months gives a minimum estimate of likely interactions in the weeks following the March visit. For calculations of mate proximity for the 1997 breeding season only data from the March 1997 visit was used.

Artificial burrows were checked every three days for tagged and untagged. Disruption of animals immediately following release was minimised by checking only on day 3 and day 10.

**Diet and food availability**

Estimates of environmental and dietary food items helped distinguish between survivorship and dispersal of tuatara related to food availability, and responses related to the availability of seabird burrows.

Environmental diversity and abundance of food items for tuatara were estimated by pitfall trapping invertebrates in each site. Tuatara are carnivorous and feed mainly on ground dwelling invertebrates, but will also eat vertebrates such as lizards and seabird eggs and chicks (Walls 1981, Ussher 1999). Pitfall trap-lines consisted of 10 '2L pitfalls' (see Appendix 2) at 10m spacing on the periphery of core release areas. Gault's solution super-saturated with sodium chloride (NaCl) provided a preservative which lasted the 2-4 months between sampling visits. Invertebrates caught were identified to Order or Family, and body length was classed as either 3-10mm, 10-20mm or >20mm. Tuatara on other islands do not eat invertebrates <3mm body length (Walls 1981, Ussher 1999) and so these were excluded from analyses of environmental invertebrate availability. Although the length of trapping periods varied considerably (range 46-141 days) between visits (but not between sites), diversity of invertebrates caught did not increase with increasing trap interval (Table 2).
Faecal pellets deposited by tuatara around frequently visited refuges were analysed to assess prey selection. Tuatara scats are difficult to find and more were found in some sites, and from some individuals. Because tuatara are density dependant foragers (Walls 1981, Ussher 1999), patterns of foraging by some individuals are assumed to be representative of others within that site. However, the number of tuatara represented by pellets is conservative because pellets cannot be attributed to known individuals with certainty.

Table 2. Relationship between the diversity of large (>10mm body length) invertebrates caught in pitfall taps and duration of trapping interval in each release site. NS = not significant (Spearman’s Rank Correlation Coefficient).

<table>
<thead>
<tr>
<th>Site</th>
<th>$r^2$</th>
<th>n</th>
<th>Probability</th>
</tr>
</thead>
<tbody>
<tr>
<td>early, even</td>
<td>0.343</td>
<td>6</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>early, clumped</td>
<td>0.771</td>
<td>6</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>mid, even</td>
<td>-0.171</td>
<td>6</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>mid, clumped</td>
<td>0.571</td>
<td>6</td>
<td>&gt;0.1 NS</td>
</tr>
</tbody>
</table>

Invertebrate remains in scats were identified to Order or Family and estimates of size made by comparison of fragments with whole specimens in a reference collection. Size categories followed classification of pitfall-trapped invertebrates (above).

**Statistical Analysis**

Numerical differences between sites were tested using Analysis of Variance (ANOVA). Some data were log-transformed to counter non-normality before analysis. For all statistical tests, individual monitoring visits (DATE) were applied as blocks for models. Proportional data between sites were analysed using SAS (Statistical Analysis System; SAS 1989) categorical data analysis (CATMOD): maximum likelihood ANOVA models. Complete models (including first order interactions between variables) were tested, non-significant variables successively dropped and models re-tested to identify significant variables. Zero values in data-sets were approximated using 0.0001 during analysis, meaning that differences in final p-values between approximations and actual zero values were at least one order of magnitude less than the minimum alpha level of significance set for all analyses (0.05).

**RESULTS**

**Survivorship**

Survivorship of tuatara was high for all sites throughout the study. All radio-tagged tuatara (5 of 8 founders) in all sites were alive after the critical period and in good health with no obvious signs of
starvation or debilitating injury. Additionally, all of these animals were alive throughout their re-capture history, whether it was shortly after the critical period or at the end of the study (510 days following release). Re-capture rates for non-radio-tagged tuatara were lower (overall; 50% for untagged tuatara vs. 100% for those radio-tagged), but in all sites at least one of these individuals was also re-captured after the critical period.

Overall re-capture rates did not differ significantly between vegetation type ($\chi^2 = 0.81$ df=1 p>0.05) or burrow dispersion ($\chi^2 = 0.0008$ df=1 p>0.05; Table 3). The re-capture of non-radio-tagged individuals throughout the duration of the study indicated that tuatara which were not re-captured following release may still have been present in the release areas.

Table 3. Number of individual tuatara re-captured after the 150 day 'critical period' on Moutohora.

<table>
<thead>
<tr>
<th>Site</th>
<th>Cumulative re-captures after day 150</th>
</tr>
</thead>
<tbody>
<tr>
<td>days since release</td>
<td>150</td>
</tr>
<tr>
<td>early &amp; even</td>
<td>8</td>
</tr>
<tr>
<td>early &amp; clumped</td>
<td>8</td>
</tr>
<tr>
<td>mid &amp; even</td>
<td>8</td>
</tr>
<tr>
<td>mid &amp; clumped</td>
<td>8</td>
</tr>
</tbody>
</table>

**Condition**

The condition of all tuatara fluctuated around their capture conditions (on Moutoki Is.) during the study. There was no significant difference in tuatara condition between sites grouped by either vegetation type or burrow dispersion (overlap of confidence intervals on Fig 1). Condition of individual animals within sites varied considerably (shown by the large confidence intervals on graph) and unpredictably between monitoring visits.

However, some patterns of average change in condition are common between sites. Ten days after release all tuatara had lost condition and of similar amounts relative to body size. Over the next 360 days (to spring 1997) most tuatara gained condition, although the magnitude of this gain varied greatly between sites and individual animals within sites. Between Spring 1997 and summer 1998 average condition decreased markedly but largely recovered in the subsequent visit in Autumn 1998. Over this period of universal loss of condition, rainfall at Whakatane (10 km from Moutohora) was considerably less than that received over all other sampling visits (Table 4).
Table 4. Rainfall in Whakatane township between monitoring visits to Moutohora.

<table>
<thead>
<tr>
<th>Season and Year</th>
<th>Summer '97</th>
<th>Autumn '97</th>
<th>Winter '97</th>
<th>Spring '97</th>
<th>Summer '98</th>
<th>Autumn '98</th>
</tr>
</thead>
<tbody>
<tr>
<td>Days since release</td>
<td>60</td>
<td>150</td>
<td>247</td>
<td>369</td>
<td>459</td>
<td>510</td>
</tr>
<tr>
<td>Average rainfall/day (mm +/- S.E.)</td>
<td>3.3 (1.6)</td>
<td>5.0 (1.6)</td>
<td>3.3 (1.2)</td>
<td>3.8 (0.8)</td>
<td>0.9 (0.3)</td>
<td>3.1 (1.1)</td>
</tr>
<tr>
<td>Total rain since last visit (mm)</td>
<td>166.6</td>
<td>458.6</td>
<td>317.8</td>
<td>466.4</td>
<td>80</td>
<td>163.2</td>
</tr>
</tbody>
</table>

Most (25 out of 27 (93%)) tuatara initially lost weight upon release and the subsequent level of weight change of individuals varied enormously throughout the study. Between capture from Moutoki and the first 10 days following release on Moutohora, tuatara on average lost 4.9% of their body weight (S. E. +/- 0.6%). The maximum level of weight loss from release weight by an individual was 30% (during spring/summer 1998). The maximum weight change observed between successive visits was -22% and +18%, both from the same male tuatara where weight lost during spring/summer 1998 was largely re-gained by autumn 1998.

**Animal movement**

**Dispersal**

Tuatara from sites where burrows were clumped moved significantly further from release locations than did tuatara from sites where burrows were evenly distributed (Table 5; Fig 2a-d, Fig 3). Additionally, male tuatara moved further than female tuatara within respective sites (Table 5) for most (13 out of 14) sampling occasions.

Table 5. Differences in the magnitude of dispersal of tuatara from release sites on Moutohora. Model includes first order interactions. NS = not significant; SIG = significant difference

<table>
<thead>
<tr>
<th>Variable</th>
<th>DF</th>
<th>F value</th>
<th>Probability</th>
</tr>
</thead>
<tbody>
<tr>
<td>Date</td>
<td>6</td>
<td>7.77</td>
<td>&lt;0.001 SIG</td>
</tr>
<tr>
<td>Vegetation Type</td>
<td>1</td>
<td>1.09</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Burrow Dispersion</td>
<td>1</td>
<td>51.12</td>
<td>&lt;0.001 SIG</td>
</tr>
<tr>
<td>Gender of Tuatara</td>
<td>1</td>
<td>18.71</td>
<td>&lt;0.001 SIG</td>
</tr>
<tr>
<td>Veg* Burrow</td>
<td>2</td>
<td>2.37</td>
<td>&gt;0.05 NS</td>
</tr>
<tr>
<td>Veg* Gender</td>
<td>1</td>
<td>0.06</td>
<td>&gt;0.5 NS</td>
</tr>
<tr>
<td>Burrow* Gender</td>
<td>1</td>
<td>0.81</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Error</td>
<td>129</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Figure 1.
Effect of vegetation type and dispersion of seabird burrows on change in condition of tuatara released onto Moutohora. Sample sizes given besides data-points. See text for formula for calculating corrected body weight (condition) of individuals.
2a. Early successional forest even burrows

2b. Mid successional forest even burrows

**KEY**
F5, M1 female/male tuatara without transmitter
F5, M1 female/male tuatara with transmitter

- release location - natural burrow
- release location - open ground
- release location - artificial burrow
- last known location

247 days duration individual followed since release
- core release area
- movement of female tuatara
- movement of male tuatara

Figure 2
Release format and dispersal over time of tuatara in each release site on Moutohora.
2c. Early successional forest
clumped burrows

2d. Mid successional forest
clumped burrows

Figure 2
Release format and dispersal over time of tuatara in each release site on Moutohora.
Figure 3.
Distance of male and female tuatara from release locations in even and clumped burrow sites on Moutohora Island. Number of individuals given above bars.
The general pattern of individual animal movement was away from release locations, even for those animals followed to the completion of the study (Appendix 3.1). However, tuatara released into sites with burrows at even distributions appear more likely to curtail outward movements (7 out of 12 individuals with no relationship between distance moved and time) than those released into sites with clumped burrows (1 out of 10 without significant correlation; Appendix 3.1).

During both breeding seasons (March 1997 & 1998) male tuatara clearly visited burrows recently visited by female tuatara and traced similar routes over a number of days. Female tuatara did not consistently visit burrows recently used by male tuatara.

**Site attachment**

In general, tuatara did not remain in similar locations between visits and the lack of specific site affinity was related to neither vegetation type nor burrow dispersion (Table 6). Average movements between visits for female tuatara were approximately 10m (Fig 4), but were significantly greater for male tuatara which moved considerably greater distances in clumped burrow areas (on average 10m more) than in even burrow sites.

<table>
<thead>
<tr>
<th>Variable</th>
<th>DF</th>
<th>F value</th>
<th>Probability</th>
</tr>
</thead>
<tbody>
<tr>
<td>Date</td>
<td>5</td>
<td>1.49</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Vegetation Type</td>
<td>1</td>
<td>1.86</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Burrow Dispersion</td>
<td>1</td>
<td>2.73</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Gender of Tuatara</td>
<td>1</td>
<td>4.83</td>
<td>&lt;0.05 SIG</td>
</tr>
<tr>
<td>Veg* Burrow</td>
<td>1</td>
<td>0.89</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Veg* Gender</td>
<td>1</td>
<td>1.31</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Burrow* Gender</td>
<td>1</td>
<td>1.80</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Error</td>
<td>104</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Likewise, burrow dispersion had no significant effect on the range length of tuatara during monitoring visits, but vegetation type had a near significant effect (Table 7). However, there were significant differences between genders with male tuatara moving on average over twice the distance that female tuatara moved during sampling periods (Fig 5).
Figure 4.
Movement of male and female tuatara in even and clumped burrow sites between monitoring visits (long-term site fidelity). Number of individuals given above bars.
Figure 5
Movement of male and female tuatara during monitoring visits (range length). Numbers of individuals given above bars.
Table 7. Influence of site characteristics on the range length of tuatara on Moutohora. Model includes first order interactions. NS = not significant; SIG = significant difference.

<table>
<thead>
<tr>
<th>Variable</th>
<th>DF</th>
<th>F value</th>
<th>Probability</th>
</tr>
</thead>
<tbody>
<tr>
<td>Date</td>
<td>5</td>
<td>1.63</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Vegetation Type</td>
<td>1</td>
<td>3.89</td>
<td>0.0510 NS</td>
</tr>
<tr>
<td>Burrow Dispersion</td>
<td>1</td>
<td>2.71</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Gender of Tuatara</td>
<td>1</td>
<td>10.60</td>
<td>&lt;0.01 SIG</td>
</tr>
<tr>
<td>Veg* Burrow</td>
<td>1</td>
<td>0.21</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Veg* Gender</td>
<td>1</td>
<td>0.67</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Burrow* Gender</td>
<td>1</td>
<td>1.17</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Error</td>
<td>114</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Reproductive viability

Breeding behaviour was observed in all four sites during March in the first and second years following release (1997 & 1998). In each site at least one effective pair of tuatara (including male and female tuatara which were caught and identified and those which were seen but not caught) were observed either cohabiting or in very close proximity to each other. Copulation was not observed at any of the sites.

The potential reproductive viability of founders was estimated for the 1997 breeding season. Distribution of known male and female tuatara in all sites except the early-regenerating clumped site shows considerable overlap in the estimated above-ground activity ranges of effective pairs (at least one male and one female animal) between late December 1997 and late June 1997 (Figs 6a-d). Proximity to potential mates was calculated for the March 1997 visit (which samples, but does not cover, the breeding season). In total, at least 80% of the 15 females were capable of copulating during this 13-day sample of the estimated 30-60-day breeding season (Table 8). Over this sample period, all females monitored in three sites and 1 out of 4 female tuatara in the early-regenerating clumped site were within range of males.

During the 1998 breeding season, proximity of male and female tuatara were generally similar to the 1997 breeding season with female tuatara within maximum recorded male tuatara movements. Additionally, in the early-regenerating even burrow site, an unknown male tuatara was present within 30m and 32m of females F1 and F5 respectively.
6a. Early successional forest
   even burrows

6b. Mid successional forest
   even burrows

**KEY**

- release location
- movement of female tuatara (entire study)
- movement of male tuatara (entire study)
- predicted range of tuatara
- from recorded refuges - summer 1997
- + last spring 1996 record + first autumn 1997 record

Figure 6
Proximity of tuatara to potential mates (known locations + 15m likely night-time movement radius) during the 1997 breeding period (includes autumn 1997 + last known location of tuatara in previous visit and first known location in following visit).
6c. Early successional forest clumped burrows

Figure 6
Proximity of tuatara to potential mates (known locations + 15m likely night-time movement radius) during the 1997 breeding period (includes autumn 1997 + last known location of tuatara in previous visit and first known location in following visit).
Table 8. Proximity of female tuatara to the nearest male tuatara during the Autumn 1997 breeding season (March 1997). Maximum daily movements of male tuatara taken from distances travelled between consecutive days on Moutohora island after 60 days following release.

<table>
<thead>
<tr>
<th>Site</th>
<th>Female</th>
<th>Distance to closest male (m)</th>
<th>Within range of male?</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>between day-time refuges</td>
<td>night-time activity (day-time refuge + 15m)</td>
</tr>
<tr>
<td>early, even</td>
<td>1</td>
<td>36</td>
<td>51</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>32</td>
<td>47</td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>11</td>
<td>26</td>
</tr>
<tr>
<td></td>
<td>4</td>
<td>7</td>
<td>21</td>
</tr>
<tr>
<td>early, clumped</td>
<td>1</td>
<td>35</td>
<td>51</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>70</td>
<td>85</td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>105</td>
<td>120</td>
</tr>
<tr>
<td></td>
<td>4</td>
<td>110</td>
<td>135</td>
</tr>
<tr>
<td>mid, even</td>
<td>1</td>
<td>13</td>
<td>28</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>17</td>
<td>32</td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>3</td>
<td>18</td>
</tr>
<tr>
<td></td>
<td>4</td>
<td>6</td>
<td>21</td>
</tr>
<tr>
<td>mid, clumped</td>
<td>1</td>
<td>1</td>
<td>16</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>0</td>
<td>15</td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>6</td>
<td>21</td>
</tr>
</tbody>
</table>

Refuges

Artificial burrows

Patterns of artificial burrow use were similar across sites. Initially, burrow use was high (at least 3 out of 5 burrows used in each site) during the first monitoring period (up to 10 days following release) but subsequently the number of tuatara using burrows decreased (Fig. 7).

Most (18 out of 20) artificial burrows were occupied by those tuatara originally released into them. Both exceptions were in the early-regenerating forest site with clumped burrows, where a male released onto open ground occupied an artificial burrow for at least 64 days from the first night of the release, and a female which moved from her release artificial burrow to another nearby and stayed there for the remainder (455 days) of the study.
Over all sites, occupancy rates of artificial burrows were equal between genders after 3 days ($\chi^2 = 1.83$ df=1 p>0.05) and 10 days ($\chi^2 = 1.99$ df=1 p>0.05) following release. Likewise, occupancy was not significantly different for radio-tagged and non-tagged tuatara after 3 ($\chi^2 = 1.11$ df=1 p>0.05) and 10 ($\chi^2 = 0$ df=1 p>0.05) days.

Other refuges
Tuatara used a wide range of refuge types over the course of the study. Type of release location had no clear influence on the next refuge by an animal (Table 9); tuatara released into burrows (artificial and natural) were most likely to use burrows as their next refuge, as were tuatara released onto open ground. Open ground was also equally likely to be the next refuge for tuatara whether released into natural or artificial burrows, or onto open ground. No tuatara immediately moved from natural burrows or open ground to artificial burrows, although two did at later dates (see above).

Table 9. Influence of release location on next choice of refuge for tuatara fitted with transmitters (n=20) on Moutohora.

<table>
<thead>
<tr>
<th>Release location</th>
<th>First refuge choice after deserting release</th>
<th>No movement (after 10 days)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Natural Burrow</td>
<td>Artificial burrow</td>
</tr>
<tr>
<td>Natural burrow</td>
<td>2</td>
<td>0</td>
</tr>
<tr>
<td>Artificial burrow</td>
<td>3</td>
<td>0</td>
</tr>
<tr>
<td>Open ground</td>
<td>4</td>
<td>0</td>
</tr>
</tbody>
</table>

Use of open ground refuges was most common shortly after release and reduced in subsequent monitoring visits. Open ground retreats included beneath clumps of living ferns and under piles of fallen branches or leaves. Only three tuatara were observed constructing their own burrows. All other animals used either seabird burrows, rock crevices or artificial burrows.

The first natural tuatara burrow was noted 8 days following release. A female tuatara (F5; Fig 2c) in the mid-regenerating even burrow site had extended the artificial burrow in which she was released by an additional 210mm by digging under the earthen floor of the burrow ‘chamber’ further into the slope. Additional scrapings typical of tuatara digging were found under an overhanging rock 1m from this animal's artificial burrow.
The other two tuatara burrows were dug by female tuatara in the mid-regenerating clumped site. Both burrows were finished when discovered in January 1998 (460 days following release) and were not present in previous visits. The first was dug under a fallen log into a sand bank within an area used continually by the animal (F2; Fig 2d) for at least 450 days. The second tuatara burrow was dug under a flat stone, again within an area used over a long period of time (approx. 450 days) by the animal (F3; Fig 2d).

**Diet and food availability**

**Diet**

Over all sites, 132 faecal pellets from an estimated 23 of the 32 founder tuatara yielded 1625 dietary items. Although only a few pellets were found in some sites (Table 10), general prey characteristics of the diets of tuatara were consistent between sites. Only invertebrates were eaten and most of these (98.8%) were ground inhabiting; few were arboreal (0.4%) or highly mobile flighted species (0.8%). Most prey (97%) were estimated to be greater than 10mm in body length.

Within all release sites except the mid-regenerating clumped burrow site, the most common prey groups eaten were beetles of the Families Scarabaeidae and Tenebrionidae (Table 10) with ants (Family Formicidae), spiders (Order Arachnida), Myriapods (Orders Diplopoda and Chilopoda) and other beetles variously the next most frequent items eaten. Prey consumption in the mid-regenerating clumped site follows a similar pattern if the 13 ants (46% of all prey eaten in that site) consumed by one tuatara are excluded from the analysis.

<table>
<thead>
<tr>
<th>Table 10. Distribution of faecal pellets and principle prey groups in tuatara diets on Moutohora.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Early even</td>
</tr>
<tr>
<td>faecal pellets n= 21</td>
</tr>
<tr>
<td>prey items n= 161</td>
</tr>
<tr>
<td>Scarabaeidae (47%)</td>
</tr>
<tr>
<td>Tenebrionidae (22%)</td>
</tr>
<tr>
<td>Arachnida (5%)</td>
</tr>
<tr>
<td>Diplopo da (4%)</td>
</tr>
<tr>
<td>Formicidae (4%)</td>
</tr>
</tbody>
</table>
Food availability

Environmental invertebrate diversity and abundance were related to sites neither by vegetation type nor by burrow dispersion (Table 11a, b); each release site supported a significantly different diversity and abundance of invertebrate fauna. For individual release sites, the environmental diversity of large (>10mm in length) invertebrates varied predictably between sites but not between seasons for individual sites (Fig 8). The early-regenerating clumped site consistently yielded the greatest number of large invertebrate species, while species diversity was consistently low or the lowest in the mid-regenerating clumped site.

Table 11. Environmental diversity (11a) and abundance (11b) of large (>10mm) invertebrates in sites differing by vegetation type and burrow dispersion on Moutohora. Model includes first order interaction. NS = not significant; SIG = significant difference.

<table>
<thead>
<tr>
<th>Variable</th>
<th>DF</th>
<th>F value</th>
<th>Probability</th>
</tr>
</thead>
<tbody>
<tr>
<td>Date</td>
<td>5</td>
<td>4.9</td>
<td>&lt;0.001 SIG</td>
</tr>
<tr>
<td>Vegetation Type</td>
<td>1</td>
<td>23.06</td>
<td>&lt;0.001 SIG</td>
</tr>
<tr>
<td>Burrow Dispersion</td>
<td>1</td>
<td>2.35</td>
<td>&gt;0.1 NS</td>
</tr>
<tr>
<td>Veg* Burrow</td>
<td>1</td>
<td>7.72</td>
<td>&lt;0.05 SIG</td>
</tr>
<tr>
<td>Error</td>
<td>222</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Variable</th>
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<th>F value</th>
<th>Probability</th>
</tr>
</thead>
<tbody>
<tr>
<td>Date</td>
<td>6</td>
<td>14.32</td>
<td>&lt;0.0001 SIG</td>
</tr>
<tr>
<td>Vegetation Type</td>
<td>1</td>
<td>0.01</td>
<td>&gt;0.5 NS</td>
</tr>
<tr>
<td>Burrow Dispersion</td>
<td>1</td>
<td>17.66</td>
<td>&lt;0.0001 SIG</td>
</tr>
<tr>
<td>Veg* Burrow</td>
<td>1</td>
<td>18.27</td>
<td>&lt;0.0001 SIG</td>
</tr>
<tr>
<td>Error</td>
<td>260</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

By comparison, the number of large invertebrates available varied predictably by season and by individual site. In general, large invertebrates were most abundant during spring and decreased to an annual low during winter (Fig. 9). During spring and summer, large invertebrates were considerably more abundant in the early-regenerating, clumped burrow site than all other sites, but decreased during autumn and winter to levels similar to those found in other sites. Large invertebrates were always least abundant in the early-regenerating even distributed burrow site.

Prey selection

On most occasions (7 out of 8) significantly fewer prey types were eaten than were available (Fig. 10). Additionally, large invertebrates were eaten in significantly greater proportions than their relative environmental abundance (Table 12).
Figure 8
Diversity of large (>10mm body length) invertebrates at release sites on Moutohora.
Figure 9
Abundance of large (>10mm body length) invertebrates at release sites on Moutohora.
Figure 10
Diversity of large (>10mm body length) prey in the diets of tuatara and in the surrounding environment on Moutohora.
Table 12. Relative abundance of large (>10mm) prey items in diets compared to environmental availability of small (<10mm) and large prey on Moutohora. Some analyses not attempted (-) due to small sample size of droppings. NS = Not significant, SIG significant difference.

<table>
<thead>
<tr>
<th>Site</th>
<th>Season</th>
<th>Proportion prey &gt;10mm (%) (sample size)</th>
<th>Statistical significance</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Diets</td>
<td>Available</td>
</tr>
<tr>
<td>early, even</td>
<td>all seasons</td>
<td>83 (21)</td>
<td>49 (70)</td>
</tr>
<tr>
<td>early, even</td>
<td>summer 1997</td>
<td>97 (5)</td>
<td>45 (9)</td>
</tr>
<tr>
<td>early, even</td>
<td>winter 1997</td>
<td>90 (3)</td>
<td>78 (10)</td>
</tr>
<tr>
<td>early, even</td>
<td>spring 1998</td>
<td>89 (7)</td>
<td>51 (9)</td>
</tr>
<tr>
<td>early, even</td>
<td>summer 1998</td>
<td>67 (6)</td>
<td>40 (9)</td>
</tr>
<tr>
<td>early, clumped</td>
<td>all seasons</td>
<td>99 (89)</td>
<td>97 (70)</td>
</tr>
<tr>
<td>early, clumped</td>
<td>spring 1996</td>
<td>100 (3)</td>
<td>95 (10)</td>
</tr>
<tr>
<td>early, clumped</td>
<td>summer 1997</td>
<td>100 (12)</td>
<td>99 (10)</td>
</tr>
<tr>
<td>early, clumped</td>
<td>autumn 1997</td>
<td>90 (1)</td>
<td>96 (10)</td>
</tr>
<tr>
<td>early, clumped</td>
<td>winter 1997</td>
<td>99 (16)</td>
<td>100 (10)</td>
</tr>
<tr>
<td>early, clumped</td>
<td>spring 1997</td>
<td>98 (30)</td>
<td>97 (9)</td>
</tr>
<tr>
<td>early, clumped</td>
<td>summer 1998</td>
<td>98 (27)</td>
<td>89 (8)</td>
</tr>
<tr>
<td>mid, even</td>
<td>all seasons</td>
<td>99 (19)</td>
<td>53 (70)</td>
</tr>
<tr>
<td>mid, even</td>
<td>spring 1996</td>
<td>100 (2)</td>
<td>56 (10)</td>
</tr>
<tr>
<td>mid, even</td>
<td>summer 1997</td>
<td>99 (12)</td>
<td>45 (10)</td>
</tr>
<tr>
<td>mid, even</td>
<td>spring 1998</td>
<td>100 (3)</td>
<td>82 (9)</td>
</tr>
<tr>
<td>mid, even</td>
<td>summer 1998</td>
<td>100 (2)</td>
<td>60 (9)</td>
</tr>
<tr>
<td>mid, clumped</td>
<td>all seasons</td>
<td>93 (3)</td>
<td>55 (69)</td>
</tr>
<tr>
<td>mid, clumped</td>
<td>spring 1996</td>
<td>100 (1)</td>
<td>57 (10)</td>
</tr>
<tr>
<td>mid, clumped</td>
<td>summer 1997</td>
<td>92 (2)</td>
<td>51 (10)</td>
</tr>
</tbody>
</table>

DISCUSSION

Success of Moutohora re-introduction

Overall, all sites chosen for tuatara re-introduced to Moutohora were suitable for at least short-term population establishment. Rates of survival and patterns of body condition change were similar for
tuatara released into both early and mid-regenerating forest types. Likewise, short-term (within trip) and long-term (between trip) fidelity of tuatara to specific areas were similar between sites supporting evenly distributed and clumped burrows. However, the distribution of burrows did influence the distance that tuatara dispersed from release areas. Tuatara released into sites where seabird burrows were clumped dispersed significantly greater distances and were more likely to continue moving away from release locations than tuatara released into areas where burrows were evenly distributed.

The universally high level of condition loss by most tuatara during summer 1998 is of unknown significance for long-term population viability. Nothing is known of annual condition changes in tuatara in established populations. Fluctuations in animal weight of up to 22% of body mass over a couple of months may have serious physiological effects on breeding potential, but may equally represent normal patterns of weight loss and gain over dry seasons.

Even though tuatara in clumped burrow sites moved further from release areas, there is strong evidence that all release groups will be able to successfully breed. All female tuatara in all groups were within range of male tuatara in at least one of the two breeding seasons. Calculations of female proximity to males and distances that male (or female) tuatara travel, are conservative values; individuals may well be capable of moving far greater distances in search of mates. Additionally, calculations of mate proximity assume that tuatara find mates through random meetings. While courtship has been observed for tuatara (Daugherty & Cree 1990), nothing is known of how tuatara find mates. Female tuatara may attract mates using pheromones. If so, then the conservative distances of 15m used in this study to estimate the probability of mate detection will be gross underestimates of the true distance.

Also, estimates are conservative because the analysis ignores un-caught males and females, some of whom were likely to be near potential mates. Subsequent monitoring should be undertaken within 5 years to determine if the patterns of continuing dispersal of tuatara from some release areas holds. If this monitoring reveals excessive dispersal of tuatara in low density, clumped burrow sites, then future releases into such habitat should consider greater numbers of founders, augmentation of populations or retention of founders within defined areas to encourage site fidelity and restrict dispersal. Given the habits of tuatara, following all founders for even short periods of time is extremely difficult. Also, the extreme longevity of tuatara suggests that, even if founders disperse widely, chance encounters between individuals over subsequent decades may enable sufficient recruitment into even small founder populations. A better understanding of mate detection by tuatara and longer-term dispersal patterns of re-introduced populations will help managers make informed
choices about founder population size, and how and when to declare a re-introduction a success or failure.

The suitability of all release sites for tuatara survival is further enforced by diet and prey choice. Studies of dietary competition between tuatara and the introduced kiore *Rattus exulans* show that tuatara eat similar numbers of small (<10mm in length) and large invertebrate (>10mm) prey (Ussher 1999), but select for larger prey when competition (kiore) is reduced (chapter 2). Tuatara on Moutohora, where kiore are absent, almost exclusively ate large, ground-dwelling invertebrates, despite an abundance of smaller prey and consumed fewer large prey species than were available. The selection of prey by tuatara indicates that food is abundant at three release sites. Food is also assumed to be abundant in the fourth site given the comparative level of prey availability at that site.

Re-introductions can be used to trial management solutions for sites that lack perceived habitat criteria for success. This study trialed artificial burrows for tuatara. Many islands and most mainland areas lack seabird colonies and so the use of artificial seabird burrows by some tuatara during this study offers an inexpensive solution to providing near-natural refuges. Desertion of artificial burrows was high following release, but that may be because preferred natural burrows were available nearby in each release site. This study suggests that the presence of burrows may be important for preventing excessive dispersal of founders. Therefore, two hypotheses which can be tested in subsequent re-introductions are:

1. artificial burrows mimic natural burrows by reducing dispersal of released tuatara, and
2. release sites require natural or artificial burrows present for a successful re-introduction.

Tuatara proposed for Tiritiri Matangi island (Chapter 5) will test this first hypothesis. Individuals will be released into sites devoid of seabird burrows but containing artificial burrows and other natural refuges (low ferns, fallen logs and leaf litter) and their condition and dispersal compared to individuals released into a seabird colony on the island. Testing both of the above hypotheses will greatly assist refinement of release protocols for future re-introductions.

Assessing long-term re-introduction success for tuatara is an un-tested process. Department of Conservation (DoC) guidelines for assessing success (DoC 1998a) are provisional and largely based on the short-term outcomes of this re-introduction and another to Titi island (Nelson 1998). Criteria for a successful re-introduction include, for 5 years following release, the re-capture of 50% of founder adults and capture of 2-3 young bred at that location. Criteria have recently been refined to incorporate the difficulty of re-locating founders following release. Further monitoring of these first tuatara re-introductions will enable refinement of these provisional guidelines.
Managing risk and increasing efficiency for species re-introductions

Species re-introductions are often expensive (e.g. Kleiman et al. 1991), time consuming and have an uncertain, but often high, risk of failure (Griffith et al. 1989, Short et al. 1992). Performance-based goals for conservation agencies (e.g. DoC 1998b) encourage managers to choose options which minimise risk and maximise potential for success. Promotion of least risk actions also discourages the development of alternative and more efficient conservation strategies.

This is particularly pertinent when considering the merits of net versus gross conservation gain from management actions. Species re-introductions can be conducted as trials, which are one-off, un-replicated management exercises, often to proven habitats (Armstrong & McLean 1995). Alternatively, they can be designed as experiments to known or unknown habitats which aim to identify how specific habitat factor(s) influence the success or failure of the release. Trials offer less risk at less cost in the short term (greater gross financial gain) compared to experiments because the outcome can be predicted with some certainty, based on the outcomes of previous re-introductions. Experimental re-introductions require replication of release sites and an effective monitoring programme, making them inherently more expensive in the short term compared to re-introductions to proven habitats. However, experiments can identify with certainty why re-introductions fail or succeed, can identify the level of management required (e.g. predator control) to achieve conservation gains and allow confident predictions of how future re-introductions will fare in similar and dissimilar habitats (Chapter 1). Therefore, in the long term experiments can identify new habitats and more effective management strategies, increasing species security and reducing expenditure (labour, time, financial) per project and thus resulting in greater net conservation benefit.

Conservation biologists often despair at the lack of experimental procedures applied to species re-introduction and restoration projects (e.g. Armstrong et al. 1994, Caughley & Gunn 1996, Veltman 1996). However, little literature contrasts the benefits to wildlife managers of adopting conservative versus experimental approaches to wildlife management, and none offer detailed incentives that matter to managers (e.g. economic comparisons) or how these systems might work over time.

Increasing the efficiency of species re-introductions can be influenced by perceptions of success and failure. The success of a re-introduction is usually based on achieving set biological criteria - usually creating a self-maintaining population (e.g. Owen & Newman 1996, DoC 1998a). Incidental gains such as the development of new transfer technologies, enhanced public access and direct information outcomes are generally not considered indicators of success (c.f. Chapter 5). However, some authors (e.g. Southgate 1994) consider that the production of new reliable information should be the only criteria for determining re-introduction success, independent of biological success or failure. To make real progress in threatened species re-introduction theory and practice, systems for
rewarding projects which investigate critical factors determining success need to be developed to replace the current penalties which encourage conservative management (Table 13). Table 13. Possible incentives for encouraging experimentation with species re-introductions. A prerequisite for re-introductions to public access or non-access areas is the requirement for an experimental design and comprehensive monitoring programme for the re-introduction (yields reliable knowledge of why the re-introduction succeeds or fails).

<table>
<thead>
<tr>
<th>Management area</th>
<th>Incentive structure</th>
</tr>
</thead>
<tbody>
<tr>
<td>Captive breeding</td>
<td>1. Tender contract to maximise production of progeny and minimise cost</td>
</tr>
<tr>
<td>Public access areas</td>
<td>1. Site receives percentage of financial gain from attracting sponsorship/increased public visitation,</td>
</tr>
<tr>
<td></td>
<td>2. Increased promotion (for access) of site,</td>
</tr>
<tr>
<td></td>
<td>3. Support for developing infrastructure (tracks, signs, public facilities) and maintaining biological security of site, and</td>
</tr>
<tr>
<td></td>
<td>4. Commitment to continuation of species re-introductions to site at set time-frames</td>
</tr>
<tr>
<td>No public access areas</td>
<td>1. Funding for re-introductions provided from state conservation budget or from revenue from public access areas,</td>
</tr>
<tr>
<td></td>
<td>2. Increased promotion (by media) of management principles for site,</td>
</tr>
<tr>
<td></td>
<td>3. Support for developing biological security of site,</td>
</tr>
<tr>
<td></td>
<td>4. Commitment to continuation of species re-introductions to site at set time-frames</td>
</tr>
</tbody>
</table>

Revising criteria for determining success

Site selection criteria need to be flexible for different situations. Since experimentation promotes diversification of re-introduction environments and management strategies (see previous), diversification of criteria for selecting sites and evaluating success should also be encouraged (Table 14, Fig 11).

Re-introductions that are primarily for biological reasons logically set biological criteria for measuring success (e.g. Owen & Newman 1996), such as recruitment, growth or population self-sustainability after set periods following release. Moreover, re-introductions may be undertaken for a variety of biological reasons (e.g. Table 14), and the perceived importance of habitat criteria for selecting new sites differs between these objectives. For example, re-introductions of predator-vulnerable species to the mainland will preferentially select sites that are easily managed or have a history of successful predator management and those that have secure funding in place. By comparison, re-introductions that aim to spread the risk of species extinction by creating new populations will prioritise sites that can support numbers of individuals above that considered to ensure ongoing population integrity.
Figure 11. Conceptual diagram of the interactions between funding, science (knowledge of species biology) and the involvement of people in biodiversity management. Optimal species management involves all three of these factors in planning and implementation of species restoration. The omission of any one of these factors results in reductions in the efficiency of management.
A common standard by which to predict population viability is the '50/500' rule (see Simberloff 1988 for review). That principle recommends that re-introductions use a minimum of 50 founders and ensure that the destination habitat be capable of supporting 500 individuals to maintain long-term genetic integrity of the population. However, these principles are based on the population dynamics of continental and laboratory species and their relevance to New Zealand conservation biology has been questioned (Craig 1991, 1994). In New Zealand, some species (e.g. tuatara (Cree & Butler 1993) have persisted as isolated and extremely small populations for many thousands of years. Therefore, some species will require far less habitat and number of individuals required to maintain long-term population persistence than those numbers set by the 50/00 principle. Additionally, population integrity can be maintained in small populations by actively interchanging individuals between separate populations to maintain or increase local genetic diversity - the meta-population approach to species management (e.g. Craig 1994).

Re-introductions for biological reasons are directly linked to species or ecosystem conservation, but re-introductions may also be for reasons which are indirectly linked to conservation outcomes. In New Zealand, the general public want greater access to wildlife in natural settings (Mortimer et al. 1996), especially threatened species (Craig et al. 1995), and conservation policy documents (e.g. DoC 1998b) recognise the value of public support for implementing and maintaining conservation projects. The development of effective predator control has enabled management of wildlife in areas of the mainland ('mainland island' management), but few are readily accessible to the public and none are located near major population centres (Saunders 1999). Achieving increased public access and education will require site selection and evaluation criteria which are primarily people-based, not biologically-based.

For example, if public access is a core objective, then re-introduction sites near population centres and those allowing ease of public access will assume priority above selection on the basis of habitat suitability (e.g. forest type or absence of predators, Table 14). In this case, a well-designed experimental re-introduction will provide information on how best to manage threats to founders. Evaluation criteria may also differ markedly from traditional self-sustainability prerequisites. Re-introductions to public access or advocacy sites may emphasise a measurable increase in public access, education or awareness as criteria for measuring success. Alternatively, or additionally, criteria for urban re-introductions may emphasise survival of founders rather than reproduction and population size (Table 14). Population persistence could instead be achieved through augmentation and meta-population management.
Table 14. Different objectives for undertaking a re-introduction result in a different diversity and ranking of the three most important selection criteria for re-introductions to new sites and criteria used to evaluate the success of a re-introduction. A core prerequisite for all re-introductions is their planning as experimental comparisons (using either contemporaneous or retrospective controls) so that the effectiveness of the management approach at meeting the set objective(s) can be determined.

<table>
<thead>
<tr>
<th>Objective of re-introduction</th>
<th>Example(s)</th>
<th>Prioritisation of site selection criteria</th>
<th>Principle Evaluation criteria</th>
</tr>
</thead>
<tbody>
<tr>
<td>Biological</td>
<td>Restoration to mainland (if island bound)</td>
<td><strong>1.</strong> Ease of management (e.g. existing 'mainland islands')&lt;br&gt;<strong>2.</strong> Potential for maximising population growth&lt;br&gt;<strong>3.</strong> Security of funding</td>
<td>Biological self-sustainability</td>
</tr>
<tr>
<td>Increase population pool</td>
<td><strong>1.</strong> Carrying capacity of destination&lt;br&gt;<strong>2.</strong> Security of founders&lt;br&gt;<strong>3.</strong> Interactions with resident wildlife</td>
<td>Biological self-sustainability</td>
<td></td>
</tr>
<tr>
<td>Restore ecosystem function</td>
<td><strong>1.</strong> Landscape integrity/fragmentation&lt;br&gt;<strong>2.</strong> Biological complexity of destination&lt;br&gt;<strong>3.</strong> Interactions with resident wildlife</td>
<td>Biological self-sustainability</td>
<td></td>
</tr>
<tr>
<td>Public access</td>
<td>Increase public access (e.g. to urban areas)</td>
<td><strong>1.</strong> Proximity to population centre&lt;br&gt;<strong>2.</strong> Ease of public access&lt;br&gt;<strong>3.</strong> Survival of founders</td>
<td>Enhanced public support Survival of founders</td>
</tr>
<tr>
<td>Public advocacy</td>
<td>Media coverage of re-introduction</td>
<td><strong>1.</strong> Prestige of site&lt;br&gt;<strong>2.</strong> Establishment of population&lt;br&gt;<strong>3.</strong> Ease of public access</td>
<td>Enhanced public support</td>
</tr>
<tr>
<td>Cultural</td>
<td>Restore cultural links</td>
<td><strong>1.</strong> Culturally important sites&lt;br&gt;<strong>2.</strong> Survival of founders&lt;br&gt;<strong>3.</strong> Biological self-sustainability</td>
<td>Fulfilment of iwi aspirations</td>
</tr>
<tr>
<td>Financial</td>
<td>Finance wider recovery actions</td>
<td><strong>1.</strong> Access for paying clients&lt;br&gt;<strong>2.</strong> Potential for site development&lt;br&gt;<strong>3.</strong> Survival of founders</td>
<td>Economic return</td>
</tr>
</tbody>
</table>
Re-introductions also offer opportunities to fund wider recovery or conservation projects. Re-introduction sites can be selected for revenue-earning potential and success measured in financial return rather than just biological gain. As for re-introductions to public access or urban sites, biological criteria for economic re-introductions can assume lower importance and may involve only marginal population establishment or accept augmentation as the means for ensuring long-term population persistence.

Alternatively, if management costs are prohibitive on mainland areas, re-introduction can still be cost-effective by charging access fees or attracting private capital. The mainland Karori Wildlife Sanctuary, 3km from Wellington city, aims to re-introduce wildlife into an area bounded by a $2.1 million predator-proof fence. Sponsorship has financed the construction of this fence and ongoing running costs, including restoration of wildlife, following its completion will be offset by charging fees for public access (Lynch 1995).

Effective restoration programmes, especially those which include re-introduction as a management tool, require consideration of funding, scientific and public involvement issues (Figure 11). Omission of any one of these factors will weaken the potential for successfully implementing the project; without funding a project cannot be started; without science or knowledge of species biology the likelihood of management actions achieving successful outcomes are greatly diminished and without public involvement disagreement between conservation stakeholders may impede or threaten the existence of the project or the potential for the project may not be fully realised. Indeed, disagreement over the best course of action for managing the introduced kiore Rattus exulans on islands inhabited by the endemic tuatara Sphenodon punctatus show that not involving all stakeholders in management decisions can hinder broad restoration projects (Chapter 2).

Considering restoration projects in terms of all three factors is especially pertinent when planning re-introductions to public access or urban sites. There, involving people can often also provide sources of finance outside traditional state funding subsidies (e.g. Matiu I., Tiritiri Matangi I. (Galbraith 1990) and Karori Reserve (Lynch 1995)). Indeed, one method of encouraging conservation managers to invest in such re-introductions could be to set criteria for economic self-sustainability, so that funding requirements are met from sources outside state-dominated financial planning frameworks for conservation. Making some re-introductions independent of state funding restrictions (e.g. for charismatic species) could encourage a greater frequency and effectiveness of re-introductions, which could be used to simultaneously support other projects for which public access and financial return are not high priorities. Adopting flexible methods of financing re-introductions and restoration projects could also achieve new populations in areas not considered priorities by conservation managers.
SUMMARY

Correctly identifying habitat needs of species will be necessary if wildlife managers wish to exploit novel environments (islands, mainland, urban areas) and foster community support through public access and education. Testing habitat needs as well-planned experiments offer more reliable information to guide future re-introductions than that generated by trial releases or releases to similar locations. Experimentation with release environments for tuatara on Moutohora shows that short-term population survivorship and reproductive viability are unaffected by vegetation type and do not depend upon large, dense seabird colonies. Future re-introductions of tuatara and other wildlife should be designed as experiments to test declared mixes of habitat factors. This will accelerate species recovery by identifying important habitat prerequisites for re-introduction and management options for achieving these, thus refining criteria used for selecting new sites and increasing confidence in subsequent management actions.

ACKNOWLEDGEMENTS

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APPENDIX 3.1

Relationship between distance of individual tuatara from release location and time since release on Moutohora. See Figs 2a-d for diagrammatic movements of male (M) and females (F) tuatara. (Pearson’s Correlation Coefficient).

<table>
<thead>
<tr>
<th>Animal</th>
<th>Correlation co-efficient</th>
<th>n=</th>
<th>Significance</th>
<th>Duration of sampling (days)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Early, even</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>M1</td>
<td>0.801</td>
<td>40</td>
<td>&lt;0.001 SIG</td>
<td>510</td>
</tr>
<tr>
<td>M2</td>
<td>0.372</td>
<td>34</td>
<td>&gt;0.1 NS</td>
<td>510</td>
</tr>
<tr>
<td>F1</td>
<td>0.901</td>
<td>28</td>
<td>&lt;0.001 SIG</td>
<td>510</td>
</tr>
<tr>
<td>F2</td>
<td>0.353</td>
<td>34</td>
<td>&gt;0.1 NS</td>
<td>510</td>
</tr>
<tr>
<td>F3</td>
<td>0.588</td>
<td>35</td>
<td>&lt;0.001 SIG</td>
<td>510</td>
</tr>
<tr>
<td>F5</td>
<td>0.632</td>
<td>9</td>
<td>&gt;0.1 NS</td>
<td>510</td>
</tr>
<tr>
<td>Early, clumped</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>M1</td>
<td>0.685</td>
<td>38</td>
<td>&lt;0.001 SIG</td>
<td>510</td>
</tr>
<tr>
<td>M2</td>
<td>0.854</td>
<td>25</td>
<td>&lt;0.001 SIG</td>
<td>151</td>
</tr>
<tr>
<td>F1</td>
<td>0.675</td>
<td>32</td>
<td>&lt;0.001 SIG</td>
<td>510</td>
</tr>
<tr>
<td>F2</td>
<td>0.677</td>
<td>45</td>
<td>&lt;0.001 SIG</td>
<td>510</td>
</tr>
<tr>
<td>F3</td>
<td>0.527</td>
<td>25</td>
<td>&lt;0.05 SIG</td>
<td>510</td>
</tr>
<tr>
<td>Mid, even</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>M1</td>
<td>-0.003</td>
<td>41</td>
<td>&gt;0.1 NS</td>
<td>510</td>
</tr>
<tr>
<td>M2</td>
<td>0.922</td>
<td>35</td>
<td>&lt;0.001 SIG</td>
<td>459</td>
</tr>
<tr>
<td>F1</td>
<td>0.123</td>
<td>30</td>
<td>&gt;0.1 NS</td>
<td>247</td>
</tr>
<tr>
<td>F2</td>
<td>0.637</td>
<td>27</td>
<td>&lt;0.001 SIG</td>
<td>510</td>
</tr>
<tr>
<td>F3</td>
<td>-0.387</td>
<td>32</td>
<td>&gt;0.1 NS</td>
<td>369</td>
</tr>
<tr>
<td>F5</td>
<td>0.547</td>
<td>11</td>
<td>&gt;0.1 NS</td>
<td>510</td>
</tr>
<tr>
<td>Mid, clumped</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>M1</td>
<td>0.876</td>
<td>38</td>
<td>&lt;0.001 SIG</td>
<td>510</td>
</tr>
<tr>
<td>M2</td>
<td>0.445</td>
<td>39</td>
<td>&lt;0.01 SIG</td>
<td>510</td>
</tr>
<tr>
<td>F1</td>
<td>-0.264</td>
<td>30</td>
<td>&gt;0.1 NS</td>
<td>369</td>
</tr>
<tr>
<td>F2</td>
<td>0.726</td>
<td>33</td>
<td>&lt;0.001 SIG</td>
<td>510</td>
</tr>
<tr>
<td>F3</td>
<td>0.408</td>
<td>37</td>
<td>&lt;0.05 SIG</td>
<td>510</td>
</tr>
</tbody>
</table>
Recommendations for the Restoration of Threatened Species Populations

Graham Ussher
School of Environmental and Marine Sciences
University of Auckland

October 1999
CHAPTER FOUR

RECOMMENDATIONS FOR THE RESTORATION OF THREATENED SPECIES POPULATIONS
EXECUTIVE SUMMARY

Further information on the management recommendations and subject topics described in the following pages can be found in the PhD thesis by Graham Ussher (1999) entitled ‘Restoration of threatened species populations: tuatara rehabilitations and reintroductions’, University of Auckland, New Zealand.

RECOMMENDATIONS

General threatened species recovery planning

- Wherever possible, design species re-introductions as experimental tests of different options rather than trials. Encourage conservation practitioners to identify more diverse options for species recovery. For example, management options for tuatara could test:
  - the value of public access to, and impact of public access, on tuatara in natural settings,
  - value of artificial refuges in sites devoid of seabird colonies,
  - techniques for re-introducing tuatara to mainland sites, and
  - merits of using captive-reared vs. wild-caught tuatara as founders for new populations.

Management of specific tuatara populations

- Defer or modify eradication programmes for kiore so that alternative methods for restoring wildlife (including tuatara on kiore-inhabited islands) can be tested (e.g. sustained local pest control).
- Test re-introductions of select invertebrates, tuatara and lizards to islands supporting kiore.
- Add open and closed canopy regenerating forest and sites with plentiful or sparse seabird burrows to options for selecting new sites for tuatara re-introductions.
- Monitor changes in body condition, reproductive success and recruitment of tuatara populations in both regenerating and mature forest on the Chicken Islands following the eradication of kiore. Methodology should aim to complement this study by testing hypotheses of variable impact on tuatara populations by kiore between habitat types.
- Monitor re-introduced tuatara on Moutohora at 5 and 10 years following release (as set by the Tuatara Recovery Group) to determine long-term population establishment.
1. RECOMMENDATIONS FOR THE ADOPTION OF THEORY TO THREATENED SPECIES RECOVERY

1.1. Summary of findings

Often, little information on species habitat needs is available to guide managers in undertaking rehabilitation or re-introduction of populations. Conservative management approaches such as species re-introductions to similar habitats or eradication of perceived threats are management approaches which maximise the probability of restoration success. Greater efficiency of management actions and greater potential for species restoration can be achieved by testing alternatives to current approaches and determining the full range of habitat requirements of species.

Restoration programmes can be undertaken as either trials or experiments. Trials (single, one-off, non-replicated events) are the most common approach to achieving species restoration. However, trials can (1) dissuade managers from investigating alternatives to currently accepted dogma, (2) narrow management perspectives and (3) provide a misplaced sense of confidence in restoration outcomes. In contrast, planning species recovery actions as well-planned experiments offers increased efficiency and effectiveness for management actions through assessment of species habitat plasticity and the potential for diversification of conservation philosophies underlying re-introductions. Identifying testable hypotheses for the management of the tuatara (*Sphenodon spp.*), an endemic New Zealand reptile, will enable assessment of the merits of re-introductions to public access areas and the identification of environmental factors important for determining population establishment (Table 1).

Planning rehabilitation and re-introduction of species as well-designed experiments offers increased efficiency and effectiveness for management actions. Both of these traits are important for meeting current biodiversity challenges such as the potential of modified and urban environments for threatened species restoration.

*Current access to tuatara*

All of the 32 established populations of tuatara are on offshore islands which prohibit public access. The Tuatara Recovery Plan (Cree & Butler 1993) identifies advocacy and education as priorities for gaining support for tuatara recovery initiatives. Access to native wildlife in natural settings is also keenly sought after by the general public (Craig *et al.* 1995). Public access has been improved by the recent release of tuatara to Matiu (Somes) Island in Wellington's harbour, and access for public in
New Zealand’s most populated city, Auckland, is proposed with plans to re-introduced tuatara to nearby Tiritiri Matangi Island.

Table 1. General areas in which hypothesis testing will be useful for the restoration of tuatara populations and the biological significance of these areas for tuatara population establishment. Experimental options are given for testing specific management alternatives.

<table>
<thead>
<tr>
<th>Testable hypotheses</th>
<th>Biological significance</th>
<th>Experimental options</th>
</tr>
</thead>
<tbody>
<tr>
<td>Introduced mammals threaten population establishment or persistence</td>
<td>-Predation of adults, juveniles and eggs or competition with founders/ persistence of remnant populations</td>
<td>-re-introduction to sites with/without mammals 1. test presence and mechanisms of impacts 2. determine level of management required to ensure tuatara persistence</td>
</tr>
<tr>
<td>Some native birds threaten population establishment</td>
<td>-Predation/ competition &amp; long-term population persistence</td>
<td>-re-introduction in presence &amp; absence of ground-nesting birds (takahe, kiwi, weka, pukeko) &amp; morepork, kingfishers</td>
</tr>
<tr>
<td>Tuatara threaten persistence of resident native fauna</td>
<td>-Predation on resident fauna -Presence of tuatara preclude re-introduction of other fauna</td>
<td>-test effects (positive or negative) of tuatara re-introduction on resident fauna -test effects of established tuatara on re-introduced native fauna (e.g. invertebrates)</td>
</tr>
<tr>
<td>Tuatara can’t establish in early forest types or closed canopy forest</td>
<td>-Food availability -Rate of cutaneous water loss/ impact on animal condition -Thermoregulation for feeding</td>
<td>-monitor tuatara diet, feeding ecology and body condition of re-introduced tuatara in different forest types, regenerating stages of forest</td>
</tr>
<tr>
<td>Released tuatara need abundant natural burrows/ refuges to prevent excessive dispersal</td>
<td>-Dispersal of founders from release sites/ lower long-term population reproductive viability</td>
<td>-monitor tuatara dispersal in sites where refuges are frequent, infrequent, clumped and evenly distributed -test use by and dispersal from artificial refuges by founders -determining maximum distances for dispersal before separation significantly reduces reproductive potential of individual tuatara</td>
</tr>
</tbody>
</table>
Re-introduction of tuatara to mainland sites has been proposed as a major goal of the recently revised Tuatara Recovery Plan (DoC 1999) and this offers a valuable opportunity to enable access to tuatara by a much wider socio-economic cross-section of society than would be possible through less accessible island re-introductions. Intensive pest control in mainland sites (‘mainland islands’) offers an excellent opportunity to bring together restoration of tuatara to mainland sites and improve access to people by siting re-introduction areas close to or within urban areas (e.g. Karori wildlife sanctuary).

1.2. Recommendations

- Adopt experimental procedures for planning and implementation of species re-introductions. Encourage greater co-ordination between conservation providers and research institutes as one method of achieving this.

- Encourage conservation managers to identify more diverse options for achieving species recovery - such as re-introduction to areas with established public access, mainland sites (if species is currently island-bound) and even selected urban areas.
2. RECOMMENDATIONS FOR THE MANAGEMENT OF THREATENED SPECIES INCLUDING TUATARA

2.1. Dietary interactions between tuatara and kiore

2.1.1. Background

The Polynesian rat or kiore *Rattus exulans* is widely regarded as the main factor responsible for the decline or extinction of a wide range of reptiles, invertebrates and birds on the New Zealand mainland and offshore islands (see Atkinson & Moller 1990 for review). Eradication programmes for kiore have largely been centred on islands which support tuatara populations because there is strong circumstantial evidence that kiore prevent recruitment of juvenile tuatara into populations and hence that these tuatara populations are under threat of extinction. However, there is little information on the mechanism by which kiore are thought to impact on tuatara - whether by direct predation of juveniles and eggs, or food competition with young or with adults which may lower the ability of adults to successfully breed.

There is widespread acknowledgement of the need for better information on how kiore impact on tuatara and other native fauna, especially since some conservation stakeholders question the dependence on eradication as the only tool for restoring wildlife on these islands (Craig 1986, Ngatiwai 1995). Some scientists and iwi propose that alternative methods of managing kiore (Table 2) may give similar conservation outcomes to the eradication programmes currently being relied upon as the only management tool.

This study assessed the effects of indirect food competition between adult tuatara and kiore by measuring changes in the diets of adult tuatara before and after the eradication of kiore. The study also measured the intensity of the competition in two different forest types in order to test hypotheses of the Historic Impact Model (Table 2) with relation to food competition.
Table 2. Assumptions of models proposed for the impact of kiore on tuatara and other wildlife.

<table>
<thead>
<tr>
<th>Model + management</th>
<th>Core principle</th>
<th>Assumptions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Predator Model</td>
<td>past, current &amp; present impacts of kiore threaten wildlife persistence.</td>
<td>Effect of kiore greater than all other factors - food availability, habitat change, population variability</td>
</tr>
<tr>
<td>Harvest Model</td>
<td>kiore impact is only recent - impact historically regulated by human harvesting.</td>
<td>Intensity of impact is positively related to density of kiore</td>
</tr>
<tr>
<td>Historic Impact Model</td>
<td>kiore impact is most intense in the past - degree of impact reduces as forest regenerates.</td>
<td>Intensity of impact is positively related to density of kiore</td>
</tr>
</tbody>
</table>

2.1.2. Methods

The study took place on Lady Alice and Whatupuke Islands in the Chicken Islands Group, Northland. Kiore were eradicated from Whatupuke I. in October 1993, and from Lady Alice I. in September 1994. Food competition was measured between kiore and tuatara in mature forest on Whatupuke and in mature and regenerating forest on Lady Alice I. Dietary responses of adult tuatara to the removal of kiore were monitored until autumn 1997.

Adult tuatara were caught by hand at night when they were active above ground. Each individual was stomach-flushed and the food items regurgitated were identified, counted and classified according the size of whole specimens as either small (<10 mm in body length) or large (>10mm in body length).

Aspects of the diets of tuatara which were measured were:
1. Type of prey eaten - invertebrate or vertebrate; classified to Order or Family taxon
2. Number of prey eaten - minimum number estimated from body parts present
3. Size of prey eaten - classed as either small or large prey (see above)
4. Foraging success - the proportion of tuatara sampled which had food present in their stomach
5. Dietary breadth - diversity and proportion of prey groups eaten (does not take into account changes in the types of prey eaten between intervals)
Dietary overlap - similarity between the diets of tuatara before and after kiore removal (takes into account changes in specific types of prey eaten)

2.1.3. Results

Changes in the diets of tuatara following the removal of kiore indicated that kiore out-compete tuatara for food items. However, the intensity of competition for food between the two species differed between the two forest types studied (Table 3). Competition was also more intense during autumn than summer in both habitats (Table 3). These results support propositions of kiore as a factor in the decline of tuatara through competition for food. However, the results also support models that propose that the impact of kiore on wildlife is related to the level of historic, human-induced, habitat modification (the Historic Impact Model; Table 3).

Table 3. Summary of changes in environmental invertebrate abundance and tuatara dietary characteristics following the removal of kiore from two successional stages of forest on Lady Alice Island. Yes = statistically significant result from analysis (p<0.05); No = statistically non-significant result from analysis.

<table>
<thead>
<tr>
<th>Response</th>
<th>Regenerating forest</th>
<th>Mature forest</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Summer</td>
<td>Autumn</td>
</tr>
<tr>
<td>Increase in number of prey consumed?</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Increase in size of prey consumed?</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Increased foraging Success ?</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Dietary Contraction (see text for definition)</td>
<td>Yes (moderate)</td>
<td>Yes (large)</td>
</tr>
<tr>
<td>Prey switching</td>
<td>Low</td>
<td>High</td>
</tr>
</tbody>
</table>

This study looked only at food competition with adult tuatara. Competition for food between kiore and young tuatara may be less, equally or greater than that recorded for adults. Results from this study should be viewed as tentative indications of support for alternative models of impact but strong indications that kiore compete for food with tuatara. The results encourage further investigation of the
dynamics of kiore impact on tuatara and other wildlife and study of the value of islands for restoring relict populations or re-introducing species in the presence of kiore.

### 2.1.4. Recommendations

- Defer or modify eradication programmes for kiore on remaining tuatara islands to include assessment of proposed alternative approaches for managing kiore and restoring tuatara and other wildlife (e.g. sustained local pest control as is being implemented for Chevron skinks on Great Barrier Island).

- Experimentally test re-introduction of select invertebrates, tuatara and lizards to islands supporting kiore and with mature forest in order to determine the value of such islands as restoration resources.

- Monitor longer-term changes in individual tuatara condition, population structure and reproductive success on Lady Alice Island.

### 2.2. Habitat criteria for selection of sites for re-introductions of tuatara

#### 2.2.1. Background

The selection of new sites for species re-introductions is heavily determined by the perceived habitat needs of the species. If a habitat factor deemed important to species survival is absent from a proposed site (e.g. refuges, presence of potential predators), then that site is usually discarded in favour of other options (e.g. Crouchley 1994, Cree & Bulter 1993). Often however, procedures for evaluating new sites are based on only poor, or no, information of the importance of habitat factors to species survival (Griffith et al. 1989, Short et al. 1992). Testing perceived habitat needs in well designed experimental re-introductions is one method of gaining reliable information on the relative importance of an environmental factor to the successful establishment of a new population.

For tuatara, the presence of an open forest canopy (for basking and egg incubation) and large numbers of seabird burrows (as refuges and to prevent excessive post-release dispersal of founders) are thought to be important criteria for selecting new re-introduction sites. The re-introduction of
tuatara to Moutohora (Whale Island), Bay of Plenty allowed an experimental test of the importance of these variables as criteria for selecting new sites.

2.2.2. Methods

Thirty-two adult tuatara were captured from nearby Moutoki Island and released into four sites on Moutohora. Each release group comprised three males and five females. Two of the release sites were in early regenerating forest and two were in mid regenerating forest. In each of these forest pairs, one site contained seabird burrows which were abundant and evenly dispersed throughout the site, and one contained seabird burrows which were at very low densities interspersed with clumps of burrows at high densities. Therefore, comparisons could be made between two replicates of early and mid regenerating forest, and two of even and clumped burrows. Twenty of the tuatara were fitted with small radio transmitters. Tuatara were released into core areas within each site. Five of the animals were released into artificial burrows comprising a 1m length of drainage tubing (burrow tunnel) and a buried plastic bucket (burrow chamber). The tuatara were monitored for 14 months following release.

Aspects of tuatara biology which were measured following their release:
1. Mortality (number of founders surviving after a given time)
2. Body condition (body weight of individuals corrected for growth over time)
3. Dispersal (straight-line distance from release point)
4. Site attachment (straight-line distance moved from last recorded location)
5. Diet (prey eaten by analysing faecal pellets)
6. Reproduction (evidence of courtship, mating, nesting or juveniles)

2.2.3. Results

Survivorship of founders was high in all sites with at least 6 out of the 8 individuals released at each site being recaptured throughout the study. In addition to survivorship, there was also no significant difference in changes in animal condition between sites, either in relation to forest age or to burrow dispersion. The condition of animals fluctuated around their release weights throughout the study and the intensity and direction (gain or loss) of weight change differed markedly between individuals within sites. The only consistent changes in condition were over summer 1998 when almost all animals from all sites lost condition. Over this period, average daily rainfall was significantly less than for all other time periods. Most individuals that lost condition over this period regained it by the following sampling visit.
The movements of animals differed significantly between even and clumped burrow sites. Tuatara released into clumped burrow areas dispersed further from released locations than those released into even burrow sites. Additionally, a greater proportion of tuatara released into clumped burrow sites continued to move away from release locations throughout the study compared to tuatara released into even burrow sites. Therefore, tuatara released into clumped burrow sites had a tendency to move further and to keep on moving away from release locations, while those in even burrow sites did not move as far from release locations and generally stopped dispersing away from release sites by the end of the study.

There were distinct differences between genders for dispersal. Within respective sites, male tuatara generally moved 1.5-2 times (but up to 5.5 times) further from release locations than female tuatara. In general, tuatara from all sites did not remain in similar locations between visits.

Use of artificial burrows by tuatara was high even 10 days after release (at least 3 out of 5 burrows used in each site), but on subsequent monitoring visits and for the remainder of the study burrow occupancy was low (maximum of 6 out 20 burrows used in all sites). Of the six tuatara which continued to use artificial burrows, some were those originally released into those burrows, others shifted from natural burrows or other artificial burrows and took up residence, and others used artificial burrows in conjunction with nearby natural burrows.

Overall, the re-introduction to the four sites on Moutohora was considered a success in the short-term. Survival of founders was high, proximity between opposite genders was regarded as sufficient to encourage mating, and breeding behaviour was noted in all sites.

2.2.4. Recommendations

- Plan further re-introductions as experimental comparisons of different options. Recovery options for tuatara could test:
  - the value of public access to tuatara in natural settings,
  - impact of public access on tuatara in natural settings,
  - value of artificial refuges in sites devoid of seabird colonies,
  - techniques for re-introducing tuatara to mainland sites, and
  - merits of using captive-reared vs. wild-caught tuatara as founders for new populations.
3. REFERENCES


CHAPTER FIVE

TRANSLOCATION PROPOSAL FOR NORTHERN TUATARA (*Sphenodon punctatus*) TO TIRITIRI MATANGI ISLAND

OVERVIEW OF RE-INTRODUCTION PROPOSAL

The message presented throughout this thesis has been one of the benefits of embracing scientific methodology in the fledgling practice of restoration and re-introduction ecology. Experimental approaches to species recovery promise to advance ecological theory and increase management efficiency, accountability and options for species recovery. Effective conservation also depends heavily on an informed and supportive general public. Through my own experiences over 8 years of part-time work on Tiritiri Matangi Island, I can testify to the benefits of encouraging public access to rare and endangered wildlife in natural settings.

Planning re-introductions to meet all of the above management criteria will require close co-operation between scientists and managers. My primary aim with the following proposal is to demonstrate that well-planned experimental re-introductions can address multiple management issues, which, left untested, will certainly hinder the rate of restoration for species and habitats. Species recovery is influenced by biological, social and financial constraints, so it makes sense to plan re-introductions where the primary aims are not solely biologically orientated. Rather, re-introduction plans should also
seek to explore alternative and innovative management options for achieving more effective conservation. The Tiritiri Matangi tuatara proposal aims to achieve effective conservation by involving people both as advocates and as financiers of the re-introduction.

### Planning and implementation of the proposal

A management proposal written from the perspective of a sole stakeholder (e.g. scientist) is unlikely to be successfully implemented since effective, applied conservation requires co-operation between multiple stakeholders. Conservation projects are also more likely to succeed if co-ordinated by a manager(s) who has a personal stake in achieving a successful outcome (e.g. financial or personal satisfaction).

For these reasons, I undertook this project as a working re-introduction plan, not merely a re-introduction proposal. The proposal itself has benefited from discussions with resource managers (government and iwi), local community interest groups, scientists and staff of the island involved. The document heavily reflects my own scientific background, but seeks to embed this within a working framework of legislative and financial constraints, management concerns, and public aspirations and opportunities.

The re-introduction proposal was submitted for consideration to the New Zealand Department of Conservation in January 1999. The form of this version does not differ markedly from that submitted except for some small grammatical corrections and links to previous chapters of this thesis. It is written for a mixed audience of scientists, managers and general public and so the structure and language used (hopefully) does not rigidly adhere to scientific protocols of assumed knowledge and conciseness. Also, only references deemed important to issues raised are given to substantiate statements and most of these are for the benefit of the thesis examiners concerning information which has not been covered in other chapters of this thesis.

This proposal is far more thorough than most for individual species re-introductions - at least in New Zealand. Thoroughness was deemed critical for the acceptance of the proposal, because the re-introduction raises important issues of unrestricted access by people to tuatara and the unknown effects of ground-foraging birds on reptiles.

To date (August 1999), the proposal has been given the full support of the national Tuatara Recovery Group (a scientific group of mixed organisational affiliation), one iwi group and the public interest group Supporters of Tiritiri Matangi Island Inc. The proposal has also been supported in principle by
both Department of Conservation conservancies within whose jurisdiction the source and destination islands, respectively, lie. The proposal has enjoyed widespread support for its scientific approach taken and the thoroughness of the planning for implementing the project. These sentiments were also echoed by the Hauraki Maori Trust Board, which represents the interests of three of the other relevant iwi groups, but who declined to approve the project because of wider management issues between itself and DoC concerning general natural resource management. Resolution of these outstanding political issues will enable the completion of consultation with stakeholders and, hopefully, the implementation of the re-introduction at the next available release period (late 2000).
PROPOSAL FOR THE RE-INTRODUCTION OF TUATARA (Sphenodon punctatus) TO TIRITIRI MATANGI ISLAND

GRAHAM USSHER
School of Environmental and Marine Sciences
University of Auckland

January 1999
EXECUTIVE SUMMARY

This proposal provides a planning, scientific and implementation framework for the re-introduction of northern tuatara (Sphenodon punctatus punctatus) to Tiritiri Matangi.

Objectives of proposal:
1. Allow general public to view and appreciate tuatara in their natural habitat
2. Enable intensive scientific study of founder populations
3. Restore tuatara as a component of a functioning northern island ecosystem
4. Increase the geographic distribution of northern tuatara

- The release will provide valuable advocacy for tuatara and the Department of Conservation and allow the general public to be better informed of, and involved in, tuatara conservation.
- 80 northern tuatara sourced from Ruamahua-iti Island in the Aldermen’s Group will be released into four sites on Tiritiri in two releases separated by a minimum of 12 months.
- The first release in October/November 1999 will comprise 60 tuatara (39 adult females; 21 adult males) released at three sites. Of the 20 tuatara destined for each site (13 females; 7 males), 16 (11 females; 5 males) will be released immediately upon arriving on the island. The remainder of each release group will be held in a temporary captive facility prior to their release up to 4-5 days after arriving on the island, and displayed to the general public near to, but not at, their respective release sites.
- The second release is dependant on the site fidelity of tuatara released into the first three sites. If site fidelity remains high, 13 females and 7 male tuatara will be captured from Ruamahua-iti (no less than 12 months following the first transfer) and released into a public access area on Tiritiri. This area will allow the general public access to see tuatara inside burrows, basking outside burrows during the day and emerging to feed at night.
- Planning and design of the translocation reflect the scientific opportunities for assessing:
  5. aspects of tuatara habitat use, biology and the merits of greater public access to tuatara
  6. the influence of ground-feeding birds on the establishment of new tuatara populations
  7. the influence of ground-feeding birds on the likely success of threatened lizard translocations.

The project will be co-ordinated principally by the University of Auckland, and will involve iwi, DoC and a non-profit volunteer group, the Supporters of Tiritiri Matangi (SoTM) in all aspects of planning, capture and release of tuatara, and post-monitoring research opportunities.
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   1.2. TUATARA RECOVERY PLAN
   1.3. Tiritiri Matangi Island

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   2.1. Tiritiri Matangi Working Plan
   2.2. Auckland DOC Conservation Management Strategy
     2.2.1. Meeting Social Goals
     2.2.2. Meeting Biological Goals
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10.2. **IWII OF TIRITIRI MATANGI AND RUAMAHUA-ITI**
10.3. **SUPPORTERS OF TIRITIRI MATANGI (INC.)**
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SCOPE OF PROPOSAL

This proposal outlines the relative merits and describes the actions required for the (re)-introduction and successful establishment of northern tuatara (Sphenodon punctatus punctatus) to Tiritiri Matangi Island. The document is divided into a translocation proposal and an action plan. The translocation proposal covers the background, justification and discusses the method of release, while the action plan covers the practical issues and time-frames for the capture of tuatara from the source island and their subsequent release on Tiritiri.

The structure of this proposal follows that recommended in the New Zealand Department of Conservation's "Transfer guidelines for indigenous terrestrial fauna and flora". Some additional points for consideration are included which reflect the bi-partisan advocacy and conservation objectives of this proposal.

Translocation Proposal

1. INTRODUCTION

1.1. Tuatara

Tuatara (Sphenodon sp.) are a medium-sized reptile endemic to New Zealand. They are the only living representatives of the ancient Order Sphenodontida, which dates back at least 120 million years. They retain ancient anatomical and physiological features which make them of special interest internationally to evolutionary biologists. Tuatara have evolved as top carnivores in the reptile-seabird communities which characterise offshore islands (and historically mainland sites as well) of New Zealand and have evolved in an environment devoid of mammalian predators. Life history characteristics provide challenges for re-habilitation and creation of new populations of tuatara. Females reproduce on average only once every four years, offspring take at least 13 years to attain sexual maturity and adults may live for over 60 years (Cree and Butler 1993). Tuatara are predominantly nocturnal and are opportunistic foragers, feeding on a wide variety of invertebrate and vertebrate species which inhabit or use the forest floor. Daylight hours are spent sheltering in underground burrows (usually seabird burrows) or basking on the forest floor (near burrow entrances) to elevate their metabolism.
Although once widespread throughout the New Zealand mainland, successive habitat destruction and effects of introduced predators have now restricted the tuatara to approximately 30 offshore island refuges. Of these, five are relict populations comprising only a handful of, generally, aged individuals and four other populations are thought to be in decline. Many of the remaining populations are confined to extremely small islands (<5ha in area) which are vulnerable to natural catastrophe. All tuatara populations are vulnerable to accidental or deliberate introduction of mammalian predators.

Genetic studies have recognised three significant taxonomic groupings of tuatara (Daugherty et al. 1990). Gunther’s tuatara (Sphenodon guntheri) exists naturally on only one small island in the Cook Strait and is listed in the New Zealand species ranking system as a category ‘A’ species (‘requiring urgent recovery protection work’) (Molloy and Davis 1992). The other species identified (S. punctatus) is considered to comprise two sub-species - the Cook Strait tuatara (unnamed) and the northern tuatara (S. p. punctatus), both of which are listed as category ‘B’ species (‘requiring work in the short term’). Cook Strait tuatara are found on 4 islands, while northern tuatara are distributed over 26 islands covering the length of the northern half of the east coast of the North Island of New Zealand.

Each of the genotypes identified are managed within specified geographical boundaries. Tiritiri falls directly in the geographic centre of the northern tuatara’s current range and will therefore rely upon sources of only northern tuatara to re-establish a population on this island.

1.2. Tuatara Recovery Plan

Short-term (5 year) actions to maintain and enhance existing populations of tuatara are outlined in the Tuatara Recovery Plan produced by the New Zealand Department of Conservation (DoC) (Cree and Butler 1993). Particular emphasis is placed on identifying solutions to the impending extinction and suspected decline of 8 of the then 25 known populations of northern tuatara. Implementation of these actions over the subsequent 5 years (and 3 years three prior to 1993) have seen rodents and competitors removed from 6 of these islands (Red Mercury, Stanley, Cuvier, Coppermine, Lady Alice and Whatupuke Is.), successful captive breeding programmes undertaken for relict animals removed from three islands (Stanley, Cuvier and Red Mercury Is.), and re-introduction of remnant adults and their captive-born offspring to one island (Red Mercury I.).

The Recovery Plan identifies advocacy as well as biological intervention as strategies for ensuring the long-term success of tuatara recovery. It recommends that one island for each of the Cook Strait and northern tuatara be identified to meet the dual goals of increasing the current range of each subspecies and providing for increased public access to tuatara in the wild. This objective for Cook Strait
tuatara has not yet been achieved, but northern tuatara have been re-introduced to Moutohora in the Bay of Plenty in late 1996. Access to Moutohora is strictly controlled by DoC and the number of trips currently limited to approximately 6 day trips and a small number of overnight school visits each year (Matt Cook pers. comm.). Information for visitors and greater public access to these tuatara is yet to be provided. The current distribution of animals on the island does not encourage viewing or appreciation of their habitat by the public. The island also lacks the infrastructure needed for large-scale access by the public (eg. wharf, information centre, interpretative panels, shelters, high grade tracks, permanent ranger presence).

This contrasts with the recent release (October 1998) of Gunther's tuatara onto Matiu/Somes I. in Wellington's harbour. The proximity of the island to such a large population centre, a regular ferry service to the island, and unrestricted access by the general public provides a unique opportunity for public appreciation of tuatara without endangering existing populations and also for advocacy for current conservation initiatives. Although the release site was well away from public tracks, the small size of the island means that there is a high probability that animals will move to areas where the public have a greater chance of viewing them.

1.3. Tiritiri Matangi island

Tiritiri Matangi lies 4km off the Whangaparaoa Peninsula 30km north of the Auckland city centre. The island is 220 ha in area and comprises a number of small valleys perpendicular to the main ridge which runs north-west along the length of the island (Fig 1). The coast-line facing south consists of sand or pebble beaches while that of the north coast is mostly cliffs. The island is 80m at its highest point.

Maori settlement of the island and subsequent intensive pasture farming by Europeans left only small patches of forest in some of the island's gullies. Farming ceased in 1971. The goal of allowing natural regenerative processes to establish forest cover on the island was hindered by low seed germination caused by the dense cover of bracken fern (Pteridium esculentum) and grass which established following farming (West, 1982). Thus, in 1984, Tiritiri was proposed as the first major revegetation project in the Hauraki Gulf. In the intervening years over 280,000 trees and shrubs have been planted on the island with the objective of re-planting forest cover to 52% of the island's area (Ray Walter pers. comm.).

Kiore (Rattus exulans) were removed from the island in 1993 and no other introduced mammalian predators or competitors exist on the island. To date nine rare or endangered bird species have been released on the island (Table 1) and a number of plants during the re-afforestation programme but no reptile or invertebrate transfers have been attempted. The island is managed as an 'Open Sanctuary
Fig 1. Map of Tiritiri Matangi Island showing location of forest remnants, track network and infra-structure
meaning that public do not require permission to land and may arrive by either private or commercial transport, but is formally classified as a Scientific Reserve.

Tiritiri is quickly becoming world-renown as an essential destination for tourists wishing to see New Zealand's biological treasures in natural settings.

Table 1 Population status of species translocated to Tiritiri Matangi. Population estimates from R. and B. Walter. Success defined as reproduction and/or increase in population numbers since release.

<table>
<thead>
<tr>
<th>Species</th>
<th>Date of introduction</th>
<th>Estimated population size (1998)</th>
<th>Release a success or failure?</th>
</tr>
</thead>
<tbody>
<tr>
<td>Kakariki</td>
<td>1973/74/75</td>
<td>100s</td>
<td>success</td>
</tr>
<tr>
<td>N. I. Saddleback</td>
<td>1984</td>
<td>100s (800?)</td>
<td>success</td>
</tr>
<tr>
<td>Brown teal</td>
<td>1987/90</td>
<td>30</td>
<td>success</td>
</tr>
<tr>
<td>Whitehead</td>
<td>1989/90</td>
<td>100s</td>
<td>success</td>
</tr>
<tr>
<td>Takahe</td>
<td>1991/94/95</td>
<td>20</td>
<td>success</td>
</tr>
<tr>
<td>N. I. Robin</td>
<td>1992/93</td>
<td>80</td>
<td>success</td>
</tr>
<tr>
<td>Little Spotted Kiwi</td>
<td>1993/95</td>
<td>30</td>
<td>success</td>
</tr>
<tr>
<td>Stitchbird</td>
<td>1995/96</td>
<td>40-50</td>
<td>success</td>
</tr>
<tr>
<td>Kokako</td>
<td>1997/98</td>
<td>6</td>
<td>to be assessed</td>
</tr>
</tbody>
</table>

Tiritiri also has a long history of use by the Lighthouse Service. As a result, the eastern end of the island now supports a considerable number of dwellings and associated buildings (Fig 1). This 'lighthouse area' now provides accommodation for the island's three permanent staff, bunkhouse facilities for the frequent groups and overseas visitors to the island, as well as full power-generating and general maintenance and workshop facilities (Hawley 1997). A regular ferry service visits the island and many additional trips are organised by schools, clubs and societies. Visitor numbers have increased steadily to approximately 20,000 people per year (1997-1998 financial year).

Although principally managed by the Department of Conservation, a volunteer group has also contributed significant financial and practical assistance to the development of the island (Galbraith 1990). The Supporters of Tiritiri Matangi Inc. (SoTM) was formed by concerned public wishing to contribute financial and practical support directly to the island. SoTM now numbers 680 members and provides not only assistance with nursery activities and public guiding but also directly contributes to
new equipment for the island and provides financial assistance for both species transfers and research students working on the island.

Two of the island's staff have been present throughout the history of the re-vegetation project. Both of these staff also have significant personal interests in flora and fauna on the island. Their work includes monitoring of species populations and management of exotic plant pests on the island. As a result they each have detailed observations on the numbers and distribution of wildlife on the island over a number of years.

2. TIRITIRI AS A DESTINATION

A suitable planning framework for the (re)-introduction of tuatara to Tiritiri comes from four sources:

2.1. Tiritiri Matangi Working Plan

Section 4.2.2(2) of the Working Plan lists tuatara as qualifying for release on the island within the context that they are a:

'rare species that were probably part of the original fauna but whose release depends on habitat, availability of material for release and adequate security'.

Developments over the subsequent year since the production of that document have seen stronger evidence for the historical existence of tuatara on Tiritiri. Previous comparisons of the historical reptile fauna found on nearby islands has alluded to the historical presence of tuatara on the island, but analysis of midden (Maori refuse dumps) material from the island in the last year provides more compelling evidence. Tuatara bones found in a midden site on the island pre-dated the apparent settlement of the island by Maori (as indicated by the remains of kuri (native dog) and kiore (Robert Brassey pers. comm.)). Accounts by a lighthouse keeper stationed on the island in the early 1900s of handling a live tuatara (John Craig pers. comm.) also add support to the island once supporting an established population of tuatara. Probable causes of extinction for the Tiritiri population of tuatara are extensive habitat modification and the presence of kuri and kiore as predators and competitors. Annual fires were a standard part of European farm management on Tiritiri and reduced the total forest habitat to <20 ha for a sustained period (John Craig pers. comm.). Such practices would have exacerbated decline caused by kuri (Polynesian dog) and kiore, and especially increased the probability of impact by kiore by encouraging greater population fluctuations (Craig 1986, Speed 1986, Chapter 2). All three of these factors no longer act on the island.
See Section 4.5 of the proposal for information on managing human interference issues, Section 4.3 for habitat suitability and Section 6 for availability of tuatara for the release.

2.2. Auckland DoC Conservation Management Strategy

The Auckland conservancy’s objectives and intentions for the resources it administers are laid out in the Conservation Management Strategy (CMS 1995) covering 10 years from 1995-2005. A number of goals of the CMS will be met by the release of tuatara onto Tiritiri.

2.2.1. Meeting Social Goals

Section 3; Kaupapa for Auckland conservancy:
Auckland conservancy has a commitment to ‘provide opportunities for people to enjoy and learn about their heritage’

Section 4; Objectives and Implementation Provisions for Tiritiri Matangi Island
14.10.5 ‘Continue to develop Tiritiri as a focus for volunteer programs and for conservation education’

The introduction of tuatara to Tiritiri will make an important part of New Zealanders’ biological heritage accessible to the general public. The range of visitors and groups (from early school age to retired persons) which frequent the island offers a valuable opportunity for DoC to inform people from a wide cross-section of society on current conservation initiatives for tuatara.

2.2.2. Meeting Biological Goals

Section 4; Objectives and Implementation Provisions for Tiritiri Matangi Island
14.7.1 ‘Restore a thriving indigenous ecosystem representative of the Inner Gulf’

Tuatara are an integral component of functioning seabird-reptile ecosystems found on offshore islands. Restoration projects on many Inner Gulf islands seek to restore these historic communities which are now restricted to Little Barrier, Mokohinau and Noises Is. in the Inner Hauraki Gulf (as well as many other similarly inaccessible islands in the Outer Hauraki Gulf).
14.7.2 'Allow translocation of species both to and from the island as determined by a species recovery plan or an approved species translocation proposal'

14.7.3 'Undertake restoration in accordance with a Working Plan and review and update the plan regularly'

14.10.2 'Provide visitor opportunities which focus on experiencing native birds in the wild, and functioning of indigenous ecosystems....'

The need for the transfer of animals and plants to Tiritiri is recognised as part of the restoration process. Although the focus is placed on birds, there is also scope for other fauna to be made accessible to the public in a way which does not jeopardise their chances of establishment. Indeed, the current priority seemingly given to birds may be a reflection of the fact that all translocations to Tiritiri to date have been of birds. Reptiles and invertebrates are proposed as possible candidates for release on the island in the Working Plan, but have yet to be implemented.

2.3. **Tuatara Recovery Plan**

2.3.1 Three objectives of the Recovery Plan (Cree and Butler 1993) are:

- **Objective 1**: Increase the security of all existing tuatara populations
- **Objective 14**: Undertake research required for tuatara conservation
- **Objective 17**: Increase public awareness and support for tuatara conservation measures

Translocation of northern tuatara to Tiritiri will fulfil all three of the above objectives. The Recovery Plan also advocates a long-term, wider ranging objective:

**Option 6.4**

'A more ambitious option - the re-establishment of tuatara on all offshore islands where they have become extinct within historic times - is not feasible within the next five years with existing staff and financial resources...... However, this option should be left open as a possible long-term goal for future recovery plans, and should be met whenever possible for individual islands within the duration of this recovery plan.' *(Emphasis added)*

Expanding the number of populations of tuatara will spread the risk of extinction for sub-species or regional genotypes. If the population from which the Tiritiri tuatara are sourced is threatened by some event in the future (e.g. rodent invasion or natural catastrophe), these animals can act as founders to augment or re-establish the original population. However, this is reliant on the founders for Tiritiri originating from only one population so that genetic distinctiveness is retained.
The release of tuatara into new locations offers the opportunity to experimentally test the importance of different environmental factors in influencing successful establishment of founders (see Section 8 for more detail). Tiritiri offers an unparalleled opportunity to reach and inform a large number of New Zealanders and overseas visitors about tuatara in general and conservation efforts currently being undertaken. Numbers of visitors to the island are currently increasing at a general rate of 6-15% per year (based on trends from 1993-1998). For the 1996-1997 year, Tiritiri provided a unique introduction to some of New Zealand's rare and endangered wildlife for 2,649 overseas visitors (15% of total visitors (20,774) - Barbara Walter pers. comm.) indicating the island's value both as a viable tourist destination and as a tool for educating such visitors about the uniqueness and fragility of our wildlife. Additionally, over this same period 2,761 school children visited the island (16% of total visitors), and many schools have included educational visits to Tiritiri as part of their course curriculum. Because of the island's unrestricted access status, people visit for a number of reasons, not necessarily just to view wildlife. During 1996-97, 4,333 people arrived on the island by private boat (25% of total visitors). Many of these people investigate interpretative tracks and the lighthouse station area (including information centre and panels) as well as using the island's beaches. This group of visitors would benefit most from information on the uniqueness of tuatara and threats presented by introduced mammals, including illegal landings of people on wildlife reserves.

2.4. DoC Strategic Business Plan 1998 - 'Restoring the dawn chorus'

Many of the goals and objectives encapsulated within the Strategic Business Plan (DoC 1998a) can be met by a well-designed release of tuatara onto Tiritiri.

Specific expectations by public of the Department are recognised as including the need to:

- Actively manage programmes for the recovery of all threatened species to avoid their extinction where reasonable intervention could be a successful option.
- Manage special sites on the mainland and offshore islands to restore indigenous species and their habitats
- Educate people about conservation and help them to become actively involved in conservation projects.

Objectives identified for achieving the above expectations include to:

Objective 1.1.2 Enhance population numbers and distributional ranges of species and subspecies threatened with extinction, where recovery action will be effective.

Recreation Goal 3.2 Share knowledge about natural and historic heritage with visitors, to satisfy their requirements for information, increase their enjoyment and understanding of this heritage and develop an awareness of the need for its conservation.
As a subspecies, northern tuatara number some 10,000 animals covering a large number of islands. While this may imply security from extinction, individual island populations remain extremely vulnerable to exotic predator introductions. The isolation of many of these islands and the problems this presents for frequent monitoring of resident wildlife suggests that continued expansion of the range of tuatara (especially where they once were present) should be encouraged.

While priorities for conservation action for tuatara management are based on the subspecies level, management within each subspecies is firmly based on individual island populations. Recovery efforts for relict populations of northern tuatara currently focus on conserving what little genetic stock remains (Cree and Butler 1993). Emphasis on retaining the integrity of each population's uniqueness is seen as a cautionary approach given the incomplete knowledge, but continuing study, of the distinctiveness of individual populations of tuatara.

The conservation of genetic distinctiveness may therefore be seen as applicable to all individual populations, relict or otherwise. Creating new populations of tuatara is a valuable tool for increasing the number of populations of individual island stocks. The re-establishment of tuatara onto Tiritiri will increase the security and geographical range of not only northern tuatara but also that of one particular island's stock. This has already been achieved for tuatara from Moutoki I. in the Rurima Islands group. Animals from this island were used as founders for re-establishing tuatara on Moutohora (Owen & Newman 1996, Chapter 3).

The release of tuatara onto Tiritiri will provide the opportunity to bring people and tuatara closer together in a natural setting than has ever before been possible. Visitors to Tiritiri have expressed their desire for introductions of native species to continue and are prepared to pay substantially more for the opportunity to view our natural heritage in a natural environment (Mortimer 1993).

3. OBJECTIVES, OUTCOMES AND JUSTIFICATION

3.1. Objectives

The following objectives are derived from a central goal to:

Establish a self-maintaining population of northern tuatara on Tiritiri, part of which is easily accessible by the public.

The (re)-introduction of northern tuatara to Tiritiri has four objectives (in order of priority).
3.2. Outcomes and justification of objectives

3.2.1. Public Access

Outcomes

i) Allow unrestricted access for the general public to tuatara in a natural environment
ii) Make tuatara easily accessible to over one third of New Zealand’s population
iii) Enable advocacy and educational goals of the Recovery Plan to be met
iv) Foster education and support for current conservation initiatives for tuatara
v) Nurture conservation ethic and participation in conservation projects by school-children
vi) Generate financial support from sponsors for wider tuatara conservation initiatives (other than sponsored displays on Tiritiri)

Continuation of many conservation programmes is dependant on maintaining or increasing levels of public support. This may be direct through participation or indirect through financial support for individual projects or recognition of the value of conservation programmes to personal quality of life. Since conservation of biodiversity in New Zealand is reliant on tax-payer generated income and support, there is a strong incentive for managers to ensure that:

a) the profile of individual species remains high and
b) that current conservation programmes are seen to have tangible benefits for the general public.

There is a growing demand world-wide for access to wildlife in natural settings. The Tuatara Recovery Plan recommends the release of northern tuatara onto a public access island for the purpose of meeting advocacy and educational goals. The reintroduction of tuatara to Moutohora is proffered as satisfying these objectives. However, the current status of the island (see introduction) strongly suggests that very few people will get the opportunity to visit the island and fewer still will have the opportunity to either learn about or see tuatara in the wild. Unless there is a major financial

1. To allow general public to view and appreciate tuatara in their natural habitat
2. To enable intensive scientific study of founder populations
3. To restore tuatara as a component of a functioning island ecosystem
4. To increase the geographic distribution of northern tuatara
commitment to developing all aspects of Moutohora's visitor support infrastructure, the long term outcomes of the release will continue to be biologically-based rather than advocacy-based.

By contrast, Tiritiri offers the only location in New Zealand where the general public have unrestricted access to a large diversity and number of regionally and nationally threatened species in a natural environment. The island is within comfortable travelling distance to approximately one third of New Zealand's population (ca. 1 million people) and supports an infra-structure (wharf, ferry service, information centre and shop, permanent staff) capable of managing large numbers of people. Tiritiri's moderate size and well developed track network also mean that large numbers of people can be present on the island at one time but still be ensured of a unique and intimate experience with wildlife.

The centralisation of facilities at the lighthouse area means information and exhibits involving tuatara conservation will reach most visitors to the island. This provides an opportunity to display information on tuatara status, threats to populations and how the general public can act responsibly when near sensitive wildlife refuges. The wide range and number of public exposed to information about conservation activities has attracted substantial sponsorship from the commercial sector for previous species recovery work both on and off the island.

3.2.2. Scientific Study

Outcomes
i) Quality research of the relative importance of environmental factors in influencing the success of translocations
ii) Cost-efficient opportunity to conduct studies on the biology (especially breeding biology) of northern tuatara
iii) Easily accessible population of tuatara to trial new management and monitoring techniques
iv) Information on the merits of providing greater access to tuatara by the general public, including:
   a. value to the public of increased advocacy and access to tuatara
   b. effects of public access on the behaviour and physiology of tuatara

Management of tuatara populations has focused primarily on securing existing populations from decline and/or extinction. The Recovery Plan identifies research into translocation techniques and ecology of tuatara as a high priority, especially for northern tuatara (DoC 1998b). However, because all four existing carefully monitored transfers have occurred within the last 2 years, only short-term indications of success are available.
Also, because all such studies to date are driven by short, but intense, 2-3 year research programs, the commitment needed to gather longer-term, in-depth information on dispersal patterns following the release of tuatara, survivorship, breeding, and population growth may be beyond the financial and practical resources of conservation managers. The major factor contributing to these problems is the relatively isolated, and therefore expensive, working environment in which three of the four transfers have taken place (Titi, Whale and Red Mercury Is.).

Translocations of tuatara to new locations have been attempted in the past (e.g. Hislop 1920). Unfortunately, the absence of suitable monitoring programs following release means that the reasons for success or failure of these early ventures cannot be identified. The number of suitable locations for tuatara (re)-introduction are increasing dramatically as islands are freed of introduced predators and new techniques are developed for securing mainland sites from exotic predators.

The range of habitat types available for restoration of threatened fauna often exceed the range of habitats in which they are currently restricted. Criteria for selecting new sites based on current species distribution may be artificially limiting recovery options. Only a carefully designed translocation program, with a commitment to ongoing monitoring, will allow tests of the habitat needs important for the successful establishment of new tuatara populations.

Tiritiri offers the opportunity to investigate how released tuatara behave and also to build on existing research programs for tuatara (see Sections 4.1 & 8). The University of Auckland maintains a permanent ecology field-station on the island and the island's proximity to Auckland makes it an attractive and inexpensive location for researchers. Tiritiri currently maintains an average of 3-4 students on the island each year conducting research studies. Tiritiri also attracts researchers from other universities and institutions.

Additionally, current DoC staff on Tiritiri (3) have extensive field experience in ecology and research and often participate in research projects conducted on the island. Their personal interest and involvement in many aspects of wildlife monitoring on the island highlights their value in providing crucial long-term information on general species biology.

Tiritiri has an advantage over most other islands by supporting a diverse range of habitat types - from cliff communities to early through late successional forest. The planting programme on the island has seen 115 ha (52% of the island's area) planted in native forest which now provides additional habitats at varying stages of maturity. Also, approximately 75% of the island's area is inaccessible to the general public. Public access is encouraged in other areas through a well defined network of tracks and boardwalks. This provides the opportunity to test not only habitat differences but also how human
access influences patterns of tuatara habitat use and behaviour. This will be important for future translocations of tuatara to other public access islands such as that planned for Motutapu—at the entrance of Auckland’s main harbour.

3.2.3. Restoring Ecosystem Function

Outcomes

i) Increase reptile diversity on Tiritiri.

ii) Re-establish complex food chains involving reptile-invertebrate interactions which are characteristic of similar less modified island ecosystems.

Currently, Tiritiri’s ecosystems are dominated by diurnal birds. The resident reptile fauna is represented by only two diurnal species of skink, Copper skink (*Cyclodina aenea*) and Moko skink (*Leiolopisma moco*), and no geckos. Up to twelve species of reptile (including skinks, geckos and tuatara) may have once inhabited Tiritiri (Hawley 1997), indicating a significant loss of species diversity and of components important as regulators of invertebrate populations and as pollinators for many native plants.

Removal of kiore from the island has resulted in noticeable and measurable (Chris Green pers. comm.) increases in abundance of some invertebrate groups which are principle food items of tuatara on other islands. Establishing tuatara on Tiritiri will help restore missing components of complex food chains involving predator-prey relationships between reptiles and ground invertebrates.

3.2.4. Distribution of northern tuatara

Outcomes

i) Increase the number of populations of northern tuatara.

ii) Provide an island capable of supporting at least 10,000 tuatara.

iii) Increase the potential area supporting northern tuatara by 26% and the potential number of northern tuatara on islands by 15%.

Northern tuatara are found on 27 islands, 1 of which is a recent re-introduction and the rest of which are established or relict populations. All but three of these populations are now secure from introduced predators or competitors. Predator-free islands supporting northern tuatara currently total 1240 ha of area (26% of the area of all islands supporting northern tuatara) (Cree & Butler 1993).
These islands have the potential to support an estimated 77,000 animals once populations reach equilibrium. Tiritiri has the potential to support over 10,000 tuatara (see below) - and therefore offers a 15% increase in the total number of animals currently possible on rodent-free islands.

Tiritiri also meets criteria identified in the Recovery Plan for priority selection of new locations. Because final equilibrium populations of greater than 1000 individuals are considered to offer increased genetic stability, islands greater than 20 ha in size are preferential release locations (potential capacity of island calculated as 100 animals/ha over half of the available land area). Tiritiri currently has 18 ha of naturally forested habitat and 115 ha of re-planted habitat (Ray Walter pers. comm.). Additionally, some 40 ha are being allowed to revert naturally from rank grassland to forest, giving a total of 173 ha out of 220 ha which will become natural forest habitat. The remaining 47 ha is being actively managed as grassland but the use of these areas by tuatara on Stephen’s I. (Charmichael et al. 1989) suggests that these may also prove viable habitat.

4. THREATS

4.1. Predators

Four bird species present on Tiritiri have the potential to prey on tuatara at stages during their life cycle (Table 2). None of these species are considered to pose a threat to tuatara beyond the size of immature adults or to the nests of tuatara. Tuatara lay eggs in shallow nests scraped in the ground, which are then covered over with dirt and vegetation. Eggs are left to develop on their own and the young which hatch are totally independent. Juvenile tuatara are diurnal as well as nocturnal and believed to preferentially live in dense vegetation. They are at risk of predation from not only other species but also from adult tuatara.

4.1.1. Kingfisher (Halcyon sancta vagans)

Tiritiri supports approximately 12 kingfishers. Kingfishers frequently prey on small reptiles such as skinks. Skink populations on the island may still be increasing following the removal of kiore. Therefore, kingfisher numbers may well also increase with the increase in skink populations. Observations by the resident rangers suggest that kingfishers occasionally enter forested areas, but prefer grassland and coastal areas. Also, given the high numbers of resident skinks compared to the probable numbers of juvenile tuatara which may be produced by the adults released, the threat of predation by kingfishers is considered to be low.
Table 2 Degree of threat offered by potential predators of juvenile tuatara on Tiritiri Matangi.

<table>
<thead>
<tr>
<th>Potential Predator</th>
<th>Habitats Used</th>
<th>Proportion reptiles comprise diets</th>
<th>Degree of threat to tuatara</th>
</tr>
</thead>
<tbody>
<tr>
<td>Kingfisher</td>
<td>Shoreline/ grasslands</td>
<td>high</td>
<td>low</td>
</tr>
<tr>
<td>Pukeko</td>
<td>principally grassland rarely forest</td>
<td>low</td>
<td>moderate</td>
</tr>
<tr>
<td>Takahe</td>
<td>principally grassland rarely forest</td>
<td>low</td>
<td>moderate</td>
</tr>
<tr>
<td>Morepork</td>
<td>forest/ scrub</td>
<td>moderate</td>
<td>moderate</td>
</tr>
</tbody>
</table>

4.1.2. *Pukeko* (*Porphyrio porphyrio melanotus*)

Pukeko are self-introduced from the mainland and occur in moderate numbers (approximately 200) on the island. Early planted and grassland areas are the principal habitats used by pukeko on the island. Individuals have been seen on the forest margin and only very rarely in forest interiors. Pukeko will eat skinks and the chicks of other birds. Therefore, they have the potential to prey on juvenile tuatara, especially if young use open scrub or grassland areas. Many of the forest patches are surrounded by extremely thick bracken fern which offer alternative habitats and protection for juvenile tuatara from pukeko. Pukeko are considered to be a moderate threat to juvenile tuatara.

4.1.3. *Takahe* (*Porphyrio mantelli hochstetteri*)

Takahe were introduced to the island in 1991 and now number 20 individuals. Although distributed over the entire island, takahe do not use all habitats equally. Studies of their habitat use indicate that they preferentially spend the majority of their time in or on the edge of managed grassland habitats (Baber 1996) and only rarely visit forested areas. Thus, the probability of takahe encountering tuatara is small because of both spatial (habitat) and temporal (takahe are diurnal; tuatara are nocturnal and are least active over winter) differences in behaviour. Takahe will eat skinks and small birds such as quail and sparrows (Matt Baber pers. comm.), although their primary food comprises grasses and roots (Dawson 1993; Baber 1996). Takahe are therefore a moderate threat to juvenile and adult tuatara.
4.1.4. Morepork (Ninox novaeseelandiae novaeseelandiae)

Approximately 6 morepork are believed to inhabit the island. Morepork prey on a wide variety of animal foods, including lizards within the size range of post-hatchling tuatara (Ray Pierce, pers. comm.). Moreover, morepork hunt during the times when juvenile tuatara will be the most active (dusk and night-time), and predominantly use the open forested areas on the island. Analysis of the diet of morepork elsewhere indicate that most prey are either rodents (when present) or large invertebrates such as tree weta *Hemideina* spp (when rodents are absent; Ray Pierce pers. comm.; pers. obs.). Tiritiri has a large tree weta population suggesting that while predation of young tuatara by morepork is likely, there are other more abundant sources of food available. Morepork are a moderate threat to juvenile tuatara.

Morepork have significant cultural value to Ngati Paoa, an iwi of Tiritiri Matangi. If morepork are found to threaten the long-term viability of tuatara populations on Tiritiri, then management actions other than culling should be considered. Provision of morepork exclosures, retreats for juvenile tuatara or captive breeding and release of tuatara at sizes beyond morepork predation are some of the options which can be investigated.

Other bird species on the island are not considered to present a threat to either eggs, young or adult tuatara. Little spotted kiwi (*Apteryx owenii*) were released on the island in 1993 and now number approximately 30 individuals. Research indicates they use a variety of habitats but principally forested areas. L. S. kiwi on other islands eat only invertebrates and plant material (Colbourne et al. 1990). Analysis of faecal pellets from Tiritiri kiwi indicates a similar diet. No reptile remains were found in 11 faecal pellets (Sabilla Giradet pers. comm.) taken from kiwi prior to and shortly after the removal of kiore from the island.

Saddleback (*Philesturnus carunculatus rufusatell*, novaeseelanidae) and kokako (*Callaeas cinerea wilsoni*) eat live animal prey, albeit in small amounts, but are extremely unlikely to target eggs or young tuatara given the small gape size of their mouths.

4.1.5. Revegetation of Tiritiri and habitat use by birds

Availability of grassland habitat today is considerably greater that that which will be present in another 10 years. Reduction in grasslands may change the potential for impact by ground-feeding birds.
Pukeko can leave the island if favoured habitats disappear, but tahake cannot. This may result in one of two scenarios:

1. Takahe numbers increase and they use forested habitats more than at present, increasing the risk of predation to juvenile tuatara.
2. Numbers of tahake stabilise at levels that grassland habitats can support; forested habitats remain infrequently used by birds.

Takahe numbers are continuing to rise, but there are indications that competition for prime territories may be causing recent increases in adult mortality. The likely carrying capacity of Tiritiri for takahe has been estimated to be 25 birds (Baber 1996) compared to the 20 present now (it assumes habitat use consistent with current patterns). If conflict between takahe and other species (such as tuatara) is seen as an increasing likelihood, then takahe populations could be managed by removing excess birds to other locations until tuatara become established.

4.1.6. National framework for assessment of threat by ground-feeding birds

Recent releases of tuatara to other islands have been in the presence of one or more of the above potential predators. Tiritiri can be incorporated into a national research framework whereby the relative risk posed by ground-feeding birds in isolation and combination with each other to tuatara can be assessed on different islands. Islands supporting new tuatara populations, together with those tentatively planned for the near future, also support the following potential predators:

- Moutohora (kingfisher?)
- Red Mercury I. morepork, (kingfisher?)
- Mana takahe, black-backed gulls, (kingfisher & pukeko?)
- Maud I. takahe, (kingfisher & pukeko?)
- Matiu/Somes black-backed gulls, (kingfisher?)
- Tiritiri takahe, pukeko, kingfisher, morepork

4.2. Competitors

A number of birds on Tiritiri are potential competitors with tuatara. Those mentioned in Section 4.1 all eat invertebrates, to some degree. However, studies of invertebrate populations since the removal of kiore indicate that key foods of tuatara are increasing significantly and in the presence of all of the above bird species (Chris Green pers. comm.). Bird populations may also increase further following kiore removal, placing more pressure on invertebrates, but factors which regulate these populations
are more likely to be related to the availability of territories rather than the availability of invertebrate foods.

4.3. Habitat Suitability

Release sites on Tiritiri are all either forested habitats or cliff scrub similar to habitats in which they are found on established islands. Levels of invertebrate food abundance on Tiritiri are similar to or higher than those recorded for Moutohora (Chris Green unpubl. data, Graham Ussher unpubl. data), where released tuatara have generally maintained release weight and show feeding consistent with an ample supply of food. Tuatara released on Titi Island in forest habitats showed significant gains in weight (Nicola Nelson 1998).

Research following the release of tuatara to Moutohora has shown that the relative availability of underground retreats help tuatara remain within general release sites. All release sites on Tiritiri will be in areas with either natural or artificial burrows (Section 8), as well as natural crevices, holes and piles of decaying plant matter within forested sites in which tuatara can take refuge.

4.4. Disease

This transfer will be from a natural, established population directly to Tiritiri. Therefore, there is a very low chance of disease being contracted by animals during transit. Tiritiri supports only natural populations of other reptiles, and therefore there is a similarly low probability of tuatara contracting disease from other reptiles on the island.

In the unlikely event that release populations are affected by disease, contingencies for identifying the type and degree of spread should be prepared. Personnel involved in post-release monitoring of the tuatara should be advised of the appropriate handling and preservation techniques for dead tuatara. Arrangements should be made with appropriate institution(s) for the analysis of dead animals (Auckland Zoological Park is the closest centre with expertise to Tiritiri).

4.5. Human Threats

Fears are often held for the safety of wildlife, such as reptiles, in the presence of people. The Tiritiri tuatara release and that on Matiu/Somes in October 1998 has raised a number of issues with the security of tuatara and the merits of public access to them. Human interference may be from legitimate visitors to the island straying from tracks to search for tuatara, or from animal collectors.
4.5.1. Visitor interference

Tiritiri is visited by approximately 20,000 people each year and most of these are day visitors. Of those who stay overnight, only a small proportion explore the island at night and all of these are either informed of protocols for night observation of wildlife or accompanied by resident DoC staff. For the three initial release sites (see Section 8), the public will have little chance to see, and therefore interact with, tuatara during the day. Tuatara may move from their release sites into areas where the public have a greater chance of seeing them, but their secretive nature means few are likely to be seen by the public. The fourth site is designed to give the public a higher chance of seeing tuatara in their natural environment. The design of that site will clearly convey the message that tuatara are within the area, but the public are not invited within the exclosure.

Public who remain on the island overnight will have a much greater chance of seeing or hearing tuatara. The track network on the island is well developed and the generally thick forest and scrub along side encourages people to remain on these. The overwhelming majority of people who stay on the island are responsible and adhere to the regulations set by DoC.

4.5.2. Animal collectors

Collectors are generally viewed as the greatest threat to individual tuatara. However, tuatara on Tiritiri are probably under little threat of being removed from the island by organised poachers, even if poachers know the general location of the release sites.

Poachers seek to minimise risk while maximising profit. Stephen’s I. has frequently been the target of one particular tuatara collector, even though the island is permanently staffed. This is probably due to the enormous numbers of animals on the island and therefore the minimal time required to capture tuatara and the low chance of being detected. The fact that poachers have not been caught on other islands with high numbers of tuatara may be because these islands are not permanently staffed, and their actions therefore go undetected. Indeed, the recent discovery of a probable Poor Knights tuatara on mainland Tutukaka has highlighted the fact that poaching from un-staffed islands probably occurs. The frequent occurrence of marijuana crops on offshore islands testifies to the regular, illegal visitors to these sanctuaries.

Tiritiri will have a very low density of animals for many years after the transfer and the island is permanently staffed. Overnight visitors and researchers are active over a large part of the island on most nights increasing the chances of poachers being detected. More significantly, the presence of
20 tuatara in each site means that on average only 4-5 individuals will be active above ground on any one night - and this only during favourable weather conditions for foraging. Even if the person is highly trained in tuatara capture, there is little chance that they will see or even hear individuals before tuatara retreat into their underground refuges.

However, despite this, determined public may leave tracks and search for tuatara, and poachers who see Tiritiri as a viable source of tuatara may capture individuals. The risk of this happening is very small. Compared to the enormous benefits which have resulted from wildlife on Tiritiri, and other islands, being more accessible to the general public, this level of risk must be seen as acceptable.

4.6. Impact on Resident Species

There is little chance of tuatara causing significant impact on resident species. The density-dependant feeding behaviour of tuatara means that species at low densities have a lower chance of being eaten than those which are more abundant. More importantly, the low numbers of tuatara released into each site, and their extremely low population growth rate, will ensure that effects on resident species will be low for many years.

Seabird chicks and eggs comprise an important part of tuatara diets on some islands. Smaller seabirds such as Fairy Prion (*Pachyptila turtur*, not present on Tiritiri) are thought to be more susceptible to predation from tuatara because of their less aggressive tendencies compared to many of the larger seabirds. Tuatara are responsible for the mortality of approximately one third of seabird eggs and chicks on Stephen's I. (Walls 1978), where tuatara densities are the highest known (Carmichael et al. 1989). However, analysis of diet for tuatara on other islands, such as the Chicken Is. and Moutohora, suggest that rates of seabird predation are much lower. This may be related to the presence of predominantly larger seabirds on these islands (Chickens - grey-faced petrels (*Pterodroma macroptera gouldi*) and flesh-footed shearwater (*Puffinus carneipes*); Moutohora - grey-faced petrels).

4.7. Island management - mowing of tracks and firebreaks

Some tuatara on Stephens Island use pasture areas for foraging and basking. Tuatara on Tiritiri may also use tracks for basking and therefore may be at risk when tracks and firebreaks are mown by the tractor-mounted rotary slasher. Post-release monitoring will indicate the likelihood of tuatara moving to track areas, but precautions should also be made before mowing is initiated in areas where tuatara are known to reside. The noise and vibration should alert basking tuatara to the approaching tractor enabling them to retreat into adjacent scrub, but inspection of areas to be mown before mowing
commences should also be considered. Revision of mowing practices should be dependant on the short-term rates of dispersal of released tuatara, but will also need to be revised as release populations expand over time either through dispersal and/or population growth.

5. INTEGRATION WITH OTHER RECOVERY PROJECTS

5.1. Habitat zoning

A number of rare or threatened species are planned for release onto Tiritiri in the future. Future bird releases are unlikely to be jeopardised by the presence of tuatara, but those of reptiles and invertebrates may be. Division of the island into discrete release zones has not been attempted for previous translocations. This perhaps is a reflection of the extremely mobile nature of the species translocated to the island thus far. All species are birds and have the ability to leave unsuitable habitats and seek those which are more optimal. This is not the case for most reptiles and invertebrates.

All valleys on Tiritiri currently support regenerating forest at various stages of development. Most of these are scrub or early successional communities which, although suitable for the release of many forest lizards, are not the preferred choice for the release of tuatara. Six valleys support mid or late successional forest (emergent or mature coastal mixed broadleaf species) which are preferred forest sites for the release of tuatara (Fig 2). Although the number of mature habitats will increase over time (as regenerating and planted areas mature), there may well be sites supporting tuatara which are also valuable for the release of other species such as some forest lizards and invertebrates (e.g. wetapunga (Deinacrida heteracantha)). Although the intrinsic rate of increase of tuatara and many lizards proposed for Tiritiri is extremely slow (Towns & Daugherty 1994), future releases in habitats supporting tuatara should assess risks and management options in conjunction with known tuatara feeding behaviour (below).

5.2. Diet of tuatara

Tuatara are carnivorous but will eat virtually anything from the size of small (7mm length) caterpillars to large seabird chicks. Their foraging strategy is, generally, to sit and wait on the forest floor until prey moves into lunging range. Food items recorded from tuatara stomachs or from analysis of scats include; invertebrates (all types), skinks, geckos, seabird eggs and chicks, juvenile tuatara, scavenged animal material. Invertebrates make up the greatest proportion of foods eaten by tuatara, in particular those which live on, or use, the forest floor and which are large (>10mm length) and slow moving.
5.3. **Risk to reptile translocations**

There is little risk that tuatara will jeopardise the success of reptile translocations to the island. Analysis of diets on other northern islands indicate that lizards are seldom eaten. Of 78 tuatara diets examined from Lady Alice I., none contained the remains of lizards. However, lizard densities during this period were extremely low (David Towns pers. comm.), probably because of the presence of kiore. Analysis of a further 120 tuatara diets from the same area in the three years following kiore removal also showed no predation by tuatara on lizards, despite an increasing lizard population (David Towns, pers. comm.). Additionally, analysis of 108 scats from tuatara recently released onto Moutohora in the presence of large densities of skinks and nocturnal geckos yielded no remains of lizards (Chapter 3).

5.4. **Risk to invertebrate translocations**

Only arboreal invertebrates are listed as possible candidates for translocation to Tiritiri in the future. Even if release sites support tuatara, the threat tuatara pose to these species is considered to be very low (Chris Green pers. comm.) If necessary, fenced tuatara exclosures will ensure an even lower risk of predation to invertebrate populations. Moreporks, rather than tuatara, present the greatest potential threat to invertebrate translocations to the island.

For both reptile and invertebrate introductions to Tiritiri, the small number of tuatara proposed for each release site coupled with their extremely slow reproductive rate, means that the level of risk they present to other species will be very low for many decades following release.

5.5. **Use of tuatara as surrogate species**

One of the concerns identified in the Tiritiri Working Plan is the potential impact of resident ground-feeding birds, especially those listed in Section 4.1, on the success of reptile translocations. The two reptile species already present on Tiritiri have high reproductive rates even though still in the presence of these predators (Chris Green pers. comm.; observations of resident DoC staff).

Tiritiri is listed as a low priority island for the introduction of endangered reptiles. Information on the level of risk from predators is required before transfers are undertaken. This proposal offers the opportunity to assess the effects of ground-feeding birds on small, slow breeding reptiles by following the fates of post-hatchling and juvenile tuatara. It may take many years for tuatara on the island to
Fig 2. Potential release sites for tuatara
breed or for numbers of juveniles to reach detectable levels to support such research. However, recovery plans for skinks (e.g. Genus Cyclodina) are not planning releases onto islands such as Tiritiri within the next 5 years. Therefore, tuatara will be a valuable surrogate species, despite the long time-frames involved.

6. SOURCE POPULATION

6.1. Choice of island

A suitable source island will be that which supports large numbers of tuatara, is reasonably accessible for the capture party, supports an ecosystem which is relatively robust to capture activities, and, most importantly, is as close as possible to Tiritiri so that regional patterns of breeding or unknown physiological adaptations adopted by animals from the source island are maintained.

Twelve islands supporting northern tuatara can be discounted as sources on the basis of insufficient population size or priorities for animals from these islands lying in re-establishing relict populations. Four other islands (all in the Hen and Chickens Group) have either recently had kiore removed from them or currently support kiore and therefore most probably represent suppressed populations. One island (Moutoki) has been the focus of a recent removal of 32 adult tuatara to re-stock Moutohora and the remaining animals are too few in numbers (approximately 120) to attempt a harvest the size of that proposed for this re-introduction (80 adults - see below).

Of the remaining 9 islands, 4 are outside the Hauraki Gulf region (Tawhiti-Rahi and Aorangi of the Poor Knights Group to the north and Karewa and Motunau to the south; Plate 1), and should be considered lower priority than the remaining five islands lying within the Outer Hauraki Gulf region. All five of these islands are situated off the east coast of the Coromandel Peninsula and are part of two adjacent island groups - the Mercury and Aldermen Island groups (Plate 1).

Two of these islands lie within the Mercury Island group and support large and healthy populations of tuatara estimated to number in the hundreds or low thousands of individuals (Cree & Butler 1993). However, these populations may be important sources for re-stocking relict islands or re-introducing tuatara to other islands in the Mercury Group. Additionally, both islands support extremely fragile environments and one supports a threatened species vulnerable to disturbance (Middle L. tusked weta Motuweta isolata). Therefore the priority of these islands should not be Tiritiri, but rather a possible future role as a source for nearby islands.
However, the Mercury group of islands are geologically closer in age since isolation (from the mainland) than the Aldermen group (below). Middle Island can therefore be considered as a potential source island if maintaining geological integrity between source and founder islands is considered to be paramount. Removal of tuatara from Middle Island may also help secure tusked weta populations by decreasing rates of predation by tuatara. Use of either of the islands in the Mercury group will require a considerable reduction in the number and range of people involved in the transfer, because of the fragile environments on these islands. Overall, the use of island(s) in the Mercury group must take lower priority because reliable estimates of tuatara population size (a necessary precursor to translocation planning) do not exist for these islands.

This leaves three islands, all of which lie within the Aldermen Group, as potential sources of tuatara for this translocation. All three islands are listed as supporting 'hundreds' or 'low hundreds' of tuatara and are therefore superficially only marginal choices for the capture of 80 adults. Hongiora is considered to have the lowest population size of the three islands (Newman & McFadden 1990, Ussher unpubl. data). It also supports enormous numbers of seabirds, whose dense burrows on the island would make movement by capture parties difficult and potentially highly destructive. Ruamahua-iti or Ruamahua-nui are therefore the remaining potential sources. Both support large seabird colonies but at much lower densities than those recorded for Hongiora.

The choice for this proposal is Ruamahua-iti, not because of the better landing sites on the island, but because of reliable estimates of tuatara numbers on this island which greatly exceed previous observation-based approximations.

6.2. Background to Ruamahua-iti
The Aldermen group of islands lies 20km off the east coast of the Coromandel Peninsula approximately half way between Mayor I. to the south and the Mercury Islands Group to the north. The Aldermen group consists of four main islands (Middle Chain, Hongiora, Ruamahua-iti and Ruamahua-nui) varying in size from 15 to 25 ha and a number of smaller islets. Three of these islands have historically supported large populations of tuatara and all are free of introduced predators. The fourth, Middle Chain I., may support a relict population of tuatara and is now rodent-free following the removal of kiore in 1993. All of these islands have been modified by humans. Maori settled the main islands, clearing forest and planting gardens as well as constructing defensive paa sites (Moore 1973, Ussher in press; Appendix 1). Occupation of the islands was thought to be on a seasonal basis only, as fresh water is only present during the wetter months (Moore 1973). Europeans did not settle or farm the islands although fires have occurred on some islands up until
about 1935 (Court et al. 1973). Ruamahua-iti is 19 ha in area and supports a diverse flora consisting of both mature and advanced regenerating forest.

6.3. Numbers of tuatara on Ruamahua-iti

The Tuatara Recovery Plan records the population size of tuatara on the island as in the ‘hundreds’, based on observations from research parties. However, recent work by the author indicates a minimum population size of 4098 and a maximum of 5271 (calculated from lower and upper daily estimates of density/ha mark-release-recapture data; Appendix 1). Even if the lowest estimate is regarded as representative of true population size, the removal of 80 adult tuatara will not endanger the population’s ability to breed and will not jeopardise Ruamahua-iti as a future source of tuatara (e.g., for restoration of Middle Chain I.)

7. TRANSFER POPULATION

7.1. Number, age and gender ratios of founder tuatara

Tuatara are proposed for release in four sites on Tiritiri. Each site will have 20 tuatara, giving a total of 80 animals required from Ruamahua-iti.

All tuatara released on Tiritiri will be adult animals. There are five reasons for this:

1. Adult tuatara are relatively easy to sex by observation and therefore capture of required numbers of male and female animals can be achieved with a high degree of certainty. It is almost impossible to accurately sex juvenile tuatara and therefore it would not be known if the transfer population of juveniles comprised equal or unfavourably skewed proportions of either gender.

2. The population estimate for Ruamahua-iti is based primarily on the abundance of adult and sub-adult animals. Therefore removing 80 adult tuatara from the population will have little effect on the viability of the population.

3. Adult tuatara are more easily caught on Ruamahua-iti compared to juveniles (out of 176 tuatara caught by Ussher (unpubl. data), 8 were considered juveniles and 168 adults). Therefore, the amount of time required to capture the animals needed for the transfer will be less (and therefore any potential damage to the island less) than that compared to efforts needed to capture juvenile tuatara.
4. A release population of adult animals will breed sooner than that comprised of juveniles and therefore provide the fastest means for growth and expansion of release populations on Tiritiri.

5. Short-term success of transfers of adult tuatara to new islands has been demonstrated (Moutohora and Titi). The fates of juvenile tuatara released onto new islands are being studied on Matiu. Release of juveniles is not recommended for Tiritiri and should not be attempted until results from the Matiu transfer are available.

The size range by which to identify mature adult tuatara for the transfer (for both females and males) will be decided upon by DoC. Assessments of population age/size structure on Ruamahua-iti indicate that the majority of female tuatara are between 170 and 190mm SVL (89% of sample n=73) and male tuatara between 190 and 230mm SVL (85% of sample n=95; Fig 3).

The proposed gender ratio for tuatara transferred to Tiritiri is 65:35 females to males. Therefore, for each release population of 20 animals, 13 of these will be females and 7 males, giving a total of 52 females and 28 males. Tuatara are polygamous (males will mate with several females) and females are not sexually active every year. Greater numbers of female tuatara than males will help the new populations on Tiritiri grow faster than if ratios were equal. However, a large number of male tuatara are also required to ensure that offspring have a diverse genetic background.

### 7.2. Capture sites on Ruamahua-iti

Release populations on Tiritiri are designed to facilitate quality research programmes. Therefore, it is essential that each population exhibit similar size and gender characteristics (discussed above) and that capture locations on Ruamahua-iti also remain comparable. Capture areas which should be excluded from capture efforts are the highly burrowed upper catchment area and the landing beach area which contrasts with the plateau area covering much of the island. Tuatara on this beach area may exhibit different feeding behaviours (e.g. scavenging on the beach) which may influence their behaviour upon release on Tiritiri. Individual capture sites for animals should be recorded to indicate the numbers of animals removed from specific areas for future transfers and to ensure that release populations comprise tuatara with similar degrees of separation across the source island. Capturing tuatara from the widest possible area of Ruamahua-iti will also help ensure the widest possible genetic base for transferred animals.
7.3. Monitoring viability of Ruamahua-iti population

The removal of 80 adult tuatara from Ruamahua-iti should have no discernible effect on the ability of resident animals to find mates or to successfully breed. Post-capture monitoring of the population is not considered a priority and even if undertaken is unlikely to provide useful information on changes in population structure or reproductive viability. This is because current methods for estimating population size and age structure yield values with levels of precision (confidence limits; see also Table 5) which exceed the number of tuatara removed for this transfer.

8. RELEASE SITE AND DESIGN

The objective of the following release design is to provide a robust scientific framework for studying aspects of tuatara biology and ecology, as well as meeting the principle objective of increasing public access to wild tuatara.

8.1. Biological design of release

8.1.1. Potential release sites

There are 8 sites on Tiritiri which can be considered as high grade potential release areas (Fig 2). All of these have either high densities of seabird burrows (the two cliff sites) or support a diverse vegetation of late successional broadleaf forest or mid successional manuka/broadleaf forest. All of the inland sites are close to streams which run or have water present in pools throughout the year. All sites contain friable soil suitable for tuatara burrowing or nesting.

Four of eight potential sites have been chosen for this translocation proposal. These are (refer Fig 2):

1. Bush 1
2. Bush 21
3. Cliff site (overlooking Northeast Bay area)
4. Wattle Valley

The choice of these sites is dictated by three variables:

a. Size of habitat
b. Relative access by public
c. Vegetation type and availability of natural seabird burrows
Figure 3. Size distribution of male and female tuatara caught on Ruamahua-iti Island. 8 juvenile tuatara <150mm SVL and for which gender could not be determined are not included in the analysis. Males n=95, females n=73
8.1.2. Proposed release sites

The forested sites proposed above (1, 2 and 4) comprise some of the largest fragments on the island. Selection of these is based on the need for the largest possible area so that tuatara released have 1) a low chance of immediately dispersing outside these habitats but 2) the opportunity to do so over a greater period of time.

Experimental tests of the influence of vegetation type and burrow density in release sites on Moutohora indicate that the availability of burrows determines the degree of dispersal by tuatara. However, that study used sites with high and low densities of burrows and cannot comment on the movements of tuatara from release sites where natural seabird burrows are absent. Many islands undergoing restoration now, and many more planned for the future, have few or no resident burrowing seabirds. Therefore, more information is needed on the importance of burrows to newly released tuatara and whether cost-efficient alternatives can be provided in areas without natural burrows. Trials with artificial burrows on other islands for newly released tuatara indicate that only a small proportion of these are regularly used, but this has been complicated by the additional presence of nearby natural seabird burrows. Captive tuatara use artificial burrows provided for them and will also dig their own burrows. Tuatara released on Tiritiri are expected to eventually dig their own burrows. However, the main concern is that the absence of burrows will cause the newly released animals to disperse excessively from their release sites.

8.1.3. Experimental design

Tiritiri offers an opportunity to further investigate the dependence of tuatara on burrows (Fig 4). The cliff release site has high densities of natural burrows and supports pohutukawa/low coastal broadleaf scrub, while Bush 21 contains no natural burrows and supports a mix of transitional manuka and broadleaf species. Although the vegetation supported by the two sites differs, results from Moutohora indicate that gross vegetation differences have no effect on dispersal patterns, survivorship or condition of tuatara. Replication at each site is provided by individual animals, giving a sample size of 20 per site.

Tiritiri also offers an important opportunity to assess the indirect effects of public access on animal behaviour and habitat use (Fig 5). The track network on Tiritiri is designed to provide areas of varying visitor density and usage. The Bush 21 release site is designated as a research area and has no
tracks or facilities to encourage public access. Thick grasslands, scrub and bracken surrounding the valley further discourage public access. By contrast, visitor access is encouraged in Bush 1 but only along a well-defined boardwalk which meanders through a large part of the valley system (Fig 2). The entire route through the valley consists of a raised boardwalk, ensuring the boundary between this public access track and the surrounding environment (off-limits) is clearly defined. The release area in Bush 1 will be well away from public use areas of the valley, but tuatara may disperse to sites closer to the boardwalk. Both sites are devoid of natural seabird burrows. Both sites support similar vegetation (core area of mature broadleaf forest and mixed kanuka/emergent broadleaf species; periphery of regenerating scrub and replanted areas). Once again, replicates are provided by individual animals within each site.

The fourth release site, Wattle Valley, is designed primarily as an area where the public can view tuatara (see Section 8.2) but can also be used as a further comparison of the effects of public exposure on tuatara behaviour.

Figure 4 Experimental design for the proposed release sites on Tiritiri

8.2. Public Design of release

There are three levels of involvement proposed for the general public:

8.2.1. Day of release - short-term access
A characteristic of public releases on Tiritiri has been the supervised display of animals immediately prior to release. Specimen animals are taken from their holding boxes and shown to groups of public by experienced handlers. This is usually the only chance that the public ever have to see New Zealand endangered species at such close quarters. This should be an integral part of the public release day.

8.2.2. Chance encounters - infrequent access

Locations of three of the release areas (Cliff, Bush 1 and Bush 21) will be kept from public knowledge. Specific release sites in these areas will be well away from public tracks (in the case of Bush 1). However, especially for Bush 1, it is anticipated that tuatara will travel to areas where public may occasionally see them from tracks.

8.2.3. Public display site - engineered access

The fourth release site, Wattle Valley, is planned as a public display site (Fig 5). Here, the primary focus will be on providing the greatest possible opportunity for the public to view tuatara in their natural environment. The proposed design will incorporate the needs of tuatara, and the viewing public and will provide security from public interference with animals.

Proposed characteristics of the public display site are:

1. a location which already caters for high public use and is easily accessible to day visitors
2. suitable habitat for tuatara - mature or maturing forest, sunspots for basking
3. is off the main track, and can have seating/ information provided with minimal disturbance to surrounding vegetation

The site will be clearly defined by:

1. Interpretation
2. Track access and seating
3. Substantial fence separating public from release area
Security of the tuatara within this site will clearly be a major issue. The concept of the site is to provide a clear message that tuatara are likely to be within that defined location, but that public presence within that area is discouraged. A focal point at one end of the area will provide information and seating for the public but provision of a fence barrier and thick scrub either side will deter further access to the site (Fig 5). The fence is a visual barrier only, not a practical barrier. Security for individual tuatara will be provided not by an encompassing fence, but by secure underground retreats. Any attempts by public to interfere with tuatara will result in animals retreating into these areas. Because the site is not fully enclosed, tuatara will have the choice of whether to stay in the release area or to move to other parts of the habitat away from public tracks and the release site itself.

The ultimate goal of this concept is to allow the general public to watch tuatara basking at their burrow entrances or to see them emerge at dusk to feed. Some artificial burrows will also be constructed to allow public to see inside the burrow chamber. Although there is a low probability of tuatara using these burrows, and therefore of the public seeing a tuatara in such a burrow during the day, visitors to Tiritiri have overwhelmingly demonstrated that it is the opportunity to see native wildlife which draws them to the island, not the guarantee of doing so.

8.3. Options for Capture

Two options are proposed for the capture of the 80 tuatara required for this translocation.
1. Capture and remove all 80 tuatara from Ruamahua-iti at the same time and release them on Tiritiri in one operation.

2. Undertake two visits to Ruamahua-iti. The first to capture 60 tuatara destined for the three non-public release sites on Tiritiri (Cliff, Bush 1 and Bush 21). Depending on the success of these releases, a decision will be made concerning a second trip to Ruamahua-iti to capture 20 tuatara and release them in the public access site (Wattle Valley). Minimum time between the first and second captures would be 12 months.

The preferred option for this proposal is Option 2. This will allow an assessment of the success of the first translocation and a decision on the future of the public access scheme. If tuatara released into the first three sites disperse excessively from their release areas, then the objectives of the public access site may be difficult to fulfil. If tuatara from the first translocation show strong site fidelity after 10-12 months, then there is a strong possibility that this pattern will be repeated in the public access site, making it a viable option (Table 3).

### 8.4. Options for release

Three options exist for the first release of tuatara onto Tiritiri (1999):

1. Transfer from Ruamahua-iti and immediate release
2. Transfer from Ruamahua-iti. Hold all tuatara until the public release day
3. Transfer from Ruamahua-iti, release most animals immediately, retain specimens for public display on the release day.

The preferred option for this proposal is option 3.

This translocation is partly based around the requirement for a public release of tuatara. Public releases must be planned in advance for a specific date. Capture of tuatara from Ruamahua-iti is dependant on weather conditions and may therefore not be complete by the date of the public release. Therefore it would be prudent to capture founder tuatara in advance of the public release and hold them until that date.

A disused aviary on Tiritiri is suitable for the holding of some, but not all, tuatara. It is situated within the lighthouse area and therefore can be easily patrolled while tuatara are resident. For the public release only a small number of tuatara will be required for display. While the majority can be released into their respective sites immediately upon arriving on the island, the rest can be retained in the aviary. To more effectively manage the public on the release day, it is proposed that viewing sites for
Tuatara be set up at three points (one for each release site) on roads near to, but not at, the release areas. Tuatara of each gender can then be displayed to the public present. Suggested numbers are 2 males and 2 females at each viewing site, giving a total of 12 tuatara to be held in the aviary. This design also enables the effects of handling to be assessed on dispersal and behaviour of tuatara following release - 4 animals (2 of each gender) from each release site (giving a comparison between gender and between sites).

<table>
<thead>
<tr>
<th>Capture trip</th>
<th>Release format</th>
<th>No. tuatara</th>
<th>Participants</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 (1999) - 60 tuatara</td>
<td>immediate release</td>
<td>48</td>
<td>representatives of stakeholders, VIPs</td>
</tr>
<tr>
<td></td>
<td>hold in aviary (4 days)</td>
<td>12</td>
<td>reps. of stakeholders, VIPs, public</td>
</tr>
<tr>
<td>2 (2000) - 20 tuatara</td>
<td>immediate release</td>
<td>20</td>
<td>iwi</td>
</tr>
</tbody>
</table>

**8.5. Gene flow between release populations**

Each release population will be isolated from each other by a minimum distance of 500m. Natural population growth and expansion over time will increase the chances of gene flow between release groups, although this may take many decades.

**9. DEFINING/MEASURING SUCCESS**

Criteria for judging the biological success of tuatara translocations have been set by the Tuatara Recovery Group as:

- **After (At) 5 years**, over a 12 month period, 50% of the adults released should be recaptured and evidence of breeding on the island obtained (capture of at least 2-3 unmarked, young animals). This would not indicate that a population has established, but that prospects for its establishment appeared good.
- **After (At) 10 years**, from a sample of 30-35 animals, the ratio of tuatara ≤ 120-180 mm (snout-vent length) SVL to total animals caught would be considered to indicate:
  - ≥ 20% population had established
  - 10-19% population has not established, but prospects for establishment appear good
cause for concern

Therefore, surveys of populations by experienced personnel should take place at 5 and 10 years following release onto Tiritiri.

There are currently no criteria for measuring the success of translocation projects in enhancing public education and appreciation of wildlife, or any benchmarks from which to provide guidelines for this project. The major goal of this project is to gain a measurable increase in public awareness of tuatara and satisfaction from increased access to them. Comprehensive public surveys must be included in the post-release monitoring schedules to gauge the social value of releasing tuatara onto public access islands.

10. PARTICIPATING PARTIES/ COMMUNICATION

The organisations listed below are essential to ensuring firstly that the transfer of tuatara is a viable option and secondly that the true value of such a transfer is recognised by the wider public. Organisations are listed in alphabetical order for convenience.

10.1. Department of Conservation

DoC is responsible for approval of this proposal from a scientific and management perspective. DoC also offers valuable experience with organising public releases of wildlife to Tiritiri and advising on public relations actions for these activities. It is envisaged that DoC will take an advisory role in the organisation of the release itself and help co-ordinate capture activities on Ruamahua-iti. All planning for activities involved with or derived from the translocation will be evaluated by DoC as the legislative body responsible for the overall project.

The capture of animals from Ruamahua-iti provides the opportunity for DoC staff to assist with and be trained in techniques for monitoring, capture, handling and data collection of tuatara. Post-release monitoring and research also provides excellent opportunities for DoC involvement.

10.2. Iwi of Tiritiri Matangi and Ruamahua-iti

Iwi who claim manawhenua over Tiritiri Matangi and Ruamahua-iti are partners in conservation management under the Treaty of Waitangi and are responsible for the approval of this proposal from
a cultural perspective. If this proposal is accepted, iwi will have the opportunity to be directly involved in both the capture of tuatara from Ruamahua-iti and the release of them on Tiritiri. Also, because this proposal is written largely from a scientific perspective, it is hoped that discussions with iwi will enable the value of this translocation to Maori to be expressed in the final form of this document.

Three iwi claim manawhenua over Tiritiri Matangi. These are Kawerau, Hauraki and Ngati Paoa. Three iwi claim manawhenua over Ruamahua-iti and these are represented by the Hauraki Trust Board.

All of the above iwi have been contacted at an early stage of the planning for this proposal. All iwi have received a draft version of this proposal for consideration of the viability of the project, and if acceptable to them, the level at which they would like to be involved with the translocation.

Ngati Paoa Trust Board have given their support for the project and would like to be involved in both the capture stage of the transfer, and certainly the release of tuatara onto Tiritiri. Comments from other iwi are expected by mid 1999.

The format proposed for the capture of tuatara from Ruamahua-iti identifies the inclusion of iwi representatives as a priority and there are at least two distinct occasions where this is possible (capture 1 (60 tuatara) and capture 2 (20 tuatara)). The format proposed for the release of tuatara onto Tiritiri will enable at least three distinct occasions where iwi can be directly involved (capture 1 - (a) initial transfer to Tiritiri and release of 48 of the 60 tuatara (b) public release of remaining 12 tuatara; capture 2 - release of 20 tuatara into public display area; see also Table 3).

10.3. Supporters of Tiritiri Matangi (Inc.)

As an independent, non-profit organisation SoTM manage funds to enhance public awareness through projects and development of Tiritiri as an open sanctuary. They also have considerable experience in attracting commercial sponsors.

SoTM will be responsible for co-ordinating funding efforts and managing funds for the aspects of the project it will finance (see Section 12 Resources). SoTM will also co-ordinate public relations activities related to the release of tuatara onto Tiritiri.
10.4. University of Auckland

Auckland University has considerable experience in planning and undertaking field expeditions and detailed planning of research requirements. University staff will co-ordinate and plan capture trip(s) to Ruamahua-iti and advise on the specific requirements of tuatara required for the Tiritiri release sites. It will also advise, in conjunction with the Tuatara Recovery Group, on the design and placement of artificial burrows and other structures required to prepare release sites on Tiritiri.

The University will assist with post-release monitoring and research of tuatara on Tiritiri and provide graduate and post-graduate students to undertake such activities where possible. Information derived from such studies will be disseminated to conservation managers to assist with managing tuatara on the island.

11. PUBLIC PERCEPTION

Anticipated outcome:

A public release of tuatara on Tiritiri and a public relations program emphasising the equal contributions of the DoC, iwi, SoTM, and the University of Auckland will attract an overwhelmingly favourable response from the general public.

Based on the experience from all other transfers of endangered species to Tiritiri, public response will be overwhelmingly favourable to the release of tuatara on the island.

Previous transfers of species have been attended by up to 300 general public. The benefits of integrating public aspirations for seeing recovery operations in progress and being in close proximity to endangered species should not be underestimated. This provides an excellent opportunity to demonstrate not only conservation in action but also to demonstrate how DoC can provide direct benefits to the public from its actions.

The transfer should therefore be planned around a public release of the tuatara. Interest by media is likely to be high and media agencies should be informed prior to the release. Media presence on the island should be encouraged and facilitated wherever possible.

12. RESOURCES
12.1. Capture and transfer of tuatara

12.1.1. Personnel

1st translocation (1999):
Preferred transport and accommodation at the Aldermen group will be a charter vessel capable of sleeping a minimum of 8 persons. Accommodation on the island itself is not recommended because of the fragile environment. Use of a boat will also increase the number of persons in the capture party from 4-5 to a maximum of 8-10 persons. Auckland University will provide the team leader (Graham Ussher) for both expeditions (1999 and 2000), but if this is not possible then DoC will provide the leader. Graham Ussher has 6 years experience working on offshore islands and 6 years experience surveying, monitoring and capturing tuatara on such islands. Other personnel will preferentially consist of DoC staff (2), iwi representatives (5), SoTM representative (1) and a media representative with experience in both wilderness work and wildlife photography.

The capture group on the island should be limited to either four (4) or five (5) persons at any one time. This will maximise the gains over time for tuatara captures and minimise the impact of human presence on the island. Members of the capture party can be rotated with those on the boat to enable all participants an opportunity to visit the island and catch tuatara. Auckland University will organise and be responsible for transport of tuatara to Tiritiri by road (from Coromandel Peninsula) and by boat (from Whangaparaoa Peninsula to Tiritiri).

2nd translocation (2000):
Preferred transport will be by DoC vessel or a small charter boat. Capture party will consist of 4-5 persons and will camp on the island for the single night required to capture the 20 tuatara. The capture party will consist of: 1 team leader (Auckland Uni); 1-2 DoC staff; 3-2 iwi representatives.

12.1.2. Equipment

Specialist equipment is needed for the capture and transport of tuatara from the source island. Auckland University and DoC will jointly plan equipment requirements for the capture trip(s).

12.1.3. Financial costs
SoTM have indicated they will cover the costs associated with the capture and transfer of tuatara from the source island. Financial aid will be limited to material and transport costs. It will not cover salaries, wages or provide enumeration to members of the capture party(s).

12.1.4. Knowledge

Both DoC and Auckland University can supply personnel with the necessary level of experience for the co-ordinating and implementation of tuatara capture trips. Auckland University will advise on the characteristics and location needed for tuatara sourced from Ruamahua-iti necessary to meet requirements for the release design and subsequent studies on Tiritiri. DoC and Auckland Zoological Park will advise on the best materials and methods for retaining and transporting tuatara.

12.2. Release and post-release monitoring of tuatara

12.2.1. Personnel

Preparation of sites on Tiritiri will be undertaken principally by DoC staff on the island with the support of volunteers and SoTM. Advice on the degree of preparation required (see Section 8) and the siting of structures will be provided by Auckland University in conjunction with DoC. Post-release monitoring will be undertaken by Auckland University, but the exact nature of monitoring and research required will be guided in part by DoC requirements.

12.2.2. Equipment

Equipment requirements for preparation of release sites will be decided upon by Auckland University and DoC. Equipment required for studies following release will be provided by the individual organisation conducting the study.

12.2.3. Financial costs

SoTM have indicated that they will cover material costs involved with preparing release sites on Tiritiri and for transporting key personnel to and from the island on the release day. SoTM will fund the production of advertising material concerning the release of tuatara on the island. SoTM will also
provide some financial assistance to research students studying the tuatara on Tiritiri following the release (as per: SoTM guidelines for assisting research).

12.2.4. Knowledge

SoTM are in the best position to co-ordinate the public relations side of the release. Planning and co-ordination of these activities, in conjunction with DoC and iwi, should therefore be the responsibility of SoTM. DoC and Auckland University have experienced personnel who can advise on the design and siting of structures required in the release areas.

12.3. Budget

12.3.1. Labour and non-consumable equipment

All personnel hours required to process and complete the translocation will be donated by the organisations involved. This includes all labour required for approving the proposal, preparation of sites on Tiritiri, travel for personnel to and from locations associated with the transfer, and for all activities associated with the capture of tuatara from the source island and their release on Tiritiri. Estimated days per person which should be budgeted for by organisations involved in the different aspects of the transfer are:

- approval of proposal/ permitting requirements (DoC) unknown
- organisation of public release day (SoTM) unknown
- capture of tuatara from Ruamahua-iti (DoC, iwi, university)
  - first transfer (Oct. 1999) 4-5 days
  - second transfer (2000-2001) 2 days
- preparation of release sites on Tiritiri (DoC, SoTM, university, iwi, volunteers)
  - non-public sites 2-3 days (supervisor only)
  - public access site 2 weeks (supervisor(s) only)
- preparation of aviary on Tiritiri (DoC, volunteers) 1-2 days
- implementation of public release day (SoTM, university, iwi, DoC) 1 day
- post-release monitoring/ research (university students) dependant on project

It is expected that non-consumable equipment resources maintained by DoC and Auckland University will be made available (within reason) for the translocation. Examples of such equipment are spot-
lighting gear for night work on islands, water containers and rodent-proof barrels if necessary, suitable tuatara transportation boxes if currently existing and equipment for taking morphometric data from transferred tuatara.

12.4. Financial costs

Cost centres and outlay for aspects of the translocation are shown below (Table 4). Note that estimates incorporate the use of volunteer labour but do not incorporate potential donation of materials by sponsors. Estimates for materials are based on retail prices not discounted or wholesale prices.

Table 4: Budget for the translocation and public release of tuatara onto Tiritiri.

<table>
<thead>
<tr>
<th>COST CENTRE</th>
<th>ACTIVITY</th>
<th>MATERIALS</th>
<th>ESTIMATED COST</th>
</tr>
</thead>
<tbody>
<tr>
<td>First capture trip</td>
<td>Boat hire for 4 days</td>
<td>Blue Fin from Whitianga @ $550/day</td>
<td>$2200.00</td>
</tr>
<tr>
<td>1999</td>
<td>Food for 8 persons, 4 days</td>
<td>calculated at $10/person/day</td>
<td>$320.00</td>
</tr>
<tr>
<td></td>
<td>Consumable equipment</td>
<td>batteries (for half of persons - other half will have rechargeable batteries provided)</td>
<td>$100.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>contingencies (lamps, tape etc)</td>
<td>$50.00</td>
</tr>
<tr>
<td></td>
<td>Transfer boxes</td>
<td>60 cardboard cat/pet boxes</td>
<td>$50.00</td>
</tr>
<tr>
<td></td>
<td></td>
<td>sterile wood chips for base?</td>
<td></td>
</tr>
<tr>
<td>Transport to</td>
<td>DoC/ University vehicle</td>
<td>200km @ $0.66/km</td>
<td>donated by uni/ DoC</td>
</tr>
<tr>
<td>Auckland</td>
<td>Ferry charter</td>
<td>passengers; iwi, DoC, uni, tuatara</td>
<td>$500.00</td>
</tr>
<tr>
<td>Transport to Tiritiri</td>
<td>catering for guests</td>
<td>15 persons (DoC, iwi)</td>
<td>$200.00</td>
</tr>
<tr>
<td></td>
<td>12 artificial burrows</td>
<td>4.5m 110mm diameter drainage coil (half round of 0.75m/ burrow) @ $1.45/burrow</td>
<td>$17.40</td>
</tr>
<tr>
<td>Aviary</td>
<td>Partitions</td>
<td>7mm plywood, support batons, top rail</td>
<td>$300.00</td>
</tr>
</tbody>
</table>

187
<table>
<thead>
<tr>
<th><strong>Preparation of non-public access sites (1999)</strong></th>
<th><strong>Public release day</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>Food for 12 aviary tuatara captive bred locusts/meal-worms</td>
<td>$ 20.00</td>
</tr>
<tr>
<td>30 artificial burrows in each of two forested sites</td>
<td>23m 110mm diameter drainage coil (half round of 0.75m/ burrow @ $1.45/burrow</td>
</tr>
<tr>
<td>200 production of pamphlets</td>
<td>$ 200.00?</td>
</tr>
<tr>
<td>200 transport for guests</td>
<td>DoC vessel (Hauturu)</td>
</tr>
<tr>
<td>200 catering for guests</td>
<td>20 persons $ 250.00</td>
</tr>
<tr>
<td>200 Interpretative panels (nursery area)</td>
<td>approx. $3000.00 based on previous signs. SoTM to advise</td>
</tr>
<tr>
<td><strong>SUB-TOTAL (1st CAPTURE TRIP)</strong></td>
<td>$4300.00</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><strong>Second capture trip (2000/2001)</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>Boat hire for 2 half days maximum $300.00/half day</td>
</tr>
<tr>
<td>Food for 4 persons, 2 days calculated at $10/person/day</td>
</tr>
<tr>
<td>Consumable equipment batteries (for half of persons - other half will have rechargeable batteries provided)</td>
</tr>
<tr>
<td>200km @ $0.66/km contingencies (lamps, tape etc)</td>
</tr>
<tr>
<td>DoC/ University vehicle donated by uni</td>
</tr>
<tr>
<td>Regular ferry service 8 persons @$25/pp</td>
</tr>
<tr>
<td>Artificial burrows (8 only - other 12 from aviary) 3m 110mm diameter drainage coil (half round of 0.75m/ burrow @ $1.45/burrow</td>
</tr>
<tr>
<td>Platform and seating</td>
</tr>
<tr>
<td>Exclosure fence</td>
</tr>
<tr>
<td>Interpretative signs</td>
</tr>
<tr>
<td><strong>SUB-TOTAL (2nd CAPTURE TRIP)</strong></td>
</tr>
</tbody>
</table>
13. MONITORING/RESEARCH/MANAGEMENT REQUIREMENTS

The release design is structured to incorporate research needs, not just post-release monitoring. It is expected that Auckland University will work closely with DoC to identify priority areas of research and for the benefits of such research to be made readily available to conservation managers. Research areas identified within the design of this translocation are:

1. Further information on habitat use in different release environments, especially with respect to the availability of natural burrow retreats and the utility of artificial burrows.
2. Burrow choice and dispersal with respect to levels of human exposure.
4. Influence of potential predators on release success (longer term for eggs and young of tuatara as well).
5. Relative merits to the general public of increased access to tuatara.

The Tuatara Recovery Group has requested that efforts be made to permanently mark all tuatara released on Tiritiri. Reliable permanent marking methods for reptiles in New Zealand are currently limited to toe-clipping, although trials with photographic identification of scale patterns is under way for some lizards. Options for marking tuatara are still being considered but are unlikely to involve toe-clipping. Implantation of non-transmitting transponder micro-chip (P.I.T. tags such as Trovan™) are the favoured option and have been successfully trialed on captive tuatara.

Management requirements for released tuatara will be determined by the success of the individual releases. Information from Moutohora suggests a high degree of flexibility of habitat use amongst adult tuatara, indicated by the high rates of survival and generally favourable responses in animal condition after 14 months following the release. The results of research studies on Tiritiri should be used to identify specific management options for individual release populations on the island. Depending on the responses of the tuatara following release, further actions may include:

1. Do nothing
2. Group release populations together at fewer sites
3. Augment with more animals from Ruamahua-iti
4. Modification of the exclosure and/or burrows

A major review of the translocation should be made 2 years after the release (see Section 9). Close contact between researchers and conservation managers should be encouraged so that issues relevant to the success of the translocation can be discussed and acted on, if necessary, before the 2-year review.

14. CONTINGENCY PLAN

Despite indications from other recent transfers of tuatara that few problems should be met, there exists the possibility that tuatara may fail to establish on the island. Whether the transfer is a success or not, well designed research programmes and regular monitoring of released animals (and later their progeny) will be critical in identifying why the release succeeds or fails.

The criteria listed for assessing the success of the translocation (Section 9) will be pivotal to determining the appropriate action after 5 and 10 year time-frames. If individual populations are considered to have established, then each release population may be augmented with additional animals to accelerate population increase or managers may feel that no further action is necessary. If populations are considered to have not established within the time-frames given for measuring success (or if there are obvious signs of this prior to these time-frames), then a reassessment of the release populations should be made. Options for action may include moving and grouping animals into more successful release sites on the island, or control where possible of agent(s) responsible for the impending failure of the translocation.

Regular communication between all participating parties (DoC, iwi, SoTM and University) will ensure that any difficulties which arise either pre- or post-translocation can be quickly identified and solutions formulated.

15. ASSESSMENT OF RISKS

The potential risks in undertaking the reintroduction of tuatara to Tiritiri are listed below (Table 5).
Table 5. Assessment of risks involved with the capture and release of tuatara onto Tiritiri.

<table>
<thead>
<tr>
<th>Risk area</th>
<th>Specific activity</th>
<th>Relative merit of undertaking activity</th>
<th>Hazard prevention/mitigation</th>
<th>Probability of hazard occurring</th>
</tr>
</thead>
<tbody>
<tr>
<td>Personnel transport</td>
<td>boat to and from Ruamahua-it to and from shore</td>
<td>boat sinks/accident, cost-effective</td>
<td>use registered charter vessel</td>
<td>very low</td>
</tr>
<tr>
<td></td>
<td>use of small boats from charter boat to shore</td>
<td>sink, drown, cost-effective, practical</td>
<td>experienced personnel to operate (G. Ussher/DoC)</td>
<td>very low</td>
</tr>
<tr>
<td></td>
<td>use at night, sink, drown</td>
<td>use at night, sink, drown, low impact on island (no camping)</td>
<td>experienced personnel to operate (G. Ussher/DoC)</td>
<td>low</td>
</tr>
<tr>
<td>Capture of tuatara</td>
<td>vehicle to Auckland working at night/boating at night</td>
<td>road accident, cost-effective (c.f. helicopter)</td>
<td>vehicle Uni/DoC + experienced driver</td>
<td>very low</td>
</tr>
<tr>
<td></td>
<td>rugged terrain</td>
<td>boat sinks/accident, cost-effective</td>
<td>use registered charter vessel</td>
<td>very low</td>
</tr>
<tr>
<td></td>
<td>accident, cliffs, damage to seabird burrows</td>
<td>accident, cliffs, damage to seabird burrows</td>
<td>training of personnel, familiarity with island, communication, correct equipment, avoid sensitive areas</td>
<td>very low - low</td>
</tr>
<tr>
<td>Transport of tuatara</td>
<td>vehicle to Auckland</td>
<td>stress, mortality, cost-effective</td>
<td>past experience (Moutohora transfer)</td>
<td>very low</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>guidelines for conditions of transfer (DoC, Auck Zoo)</td>
<td></td>
</tr>
<tr>
<td>Tuatara on Tiritiri</td>
<td>release in presence of potential threats</td>
<td>* adult mortality, significant egg/ juvenile mortality, breeding inhibition</td>
<td>previous study of predator habitat use/ diet.</td>
<td>very low or undetermined</td>
</tr>
<tr>
<td></td>
<td>release in presence of public</td>
<td>removal of animals, reduced viability of release groups</td>
<td>Experience from other transfers.</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>3 release sites kept secret. Secure underground refuges.</td>
<td>very low or unknown</td>
</tr>
</tbody>
</table>
**Overall assessment of the project (above) shows that the potential (threats) and actual (financial) costs are greatly outweighed by the likely benefits (conservation, advocacy, strengthening iwi/ DoC/ university/ public ties) of releasing tuatara onto Tiritiri.**

### 16. ACKNOWLEDGEMENTS

Ray and Barbara Walter, resident DoC rangers on Tiritiri Matangi provided their personal observations of the island’s wildlife and the information on visitors. DoC scientists David Towns and Chris Green gave advice on the release design and experimental sections of the proposal. John Craig and Robert Brassey provided information on the historical presence of tuatara on Tiritiri. Ruby Jones and John Craig read earlier drafts of the proposal.

### 17. REFERENCES


Molloy, J. and Davis, A. 1992: Setting priorities for the conservation of New Zealand's threatened plants and animals. Department of Conservation, Wellington


Ussher, G. T. 1998: Update on archaeological sites on the Aldermen islands. Tane in press.


Action Plan

18. SUMMARY OF PROPOSED SCHEDULE AND TIME-FRAMES

This section is still at the planning stage, but the essential time-frames and actions which are necessary to fulfil proposals made in this document are listed below.

18.1. Time-frames for planning and submission of translocation proposal

<table>
<thead>
<tr>
<th>Time-frame</th>
<th>Actions and Details</th>
</tr>
</thead>
<tbody>
<tr>
<td>Late October 1998</td>
<td>- consideration of proposal, discussions of viability with University/ DoC staff</td>
</tr>
<tr>
<td>Early November 1998</td>
<td>- consideration of project viability with iwi - requirement from all iwi for a draft proposal to be submitted to them for discussion</td>
</tr>
<tr>
<td>November 1998</td>
<td>- preparation of draft translocation proposal</td>
</tr>
<tr>
<td>December 1998 - January 1999</td>
<td>- distribution of draft for comment by iwi, key DoC staff, and SoTM; meeting with iwi to discuss 1) viability of transfer and 2) level of involvement desired.</td>
</tr>
<tr>
<td>January 1999</td>
<td>- submission of final draft to DoC Auckland for consideration. They will distribute copies to iwi, DoC Waikato (Ruamahua-it) and SoTM, whose approval is required before the project can be accepted or undergo further planning.</td>
</tr>
</tbody>
</table>

The release date proposed for reintroducing tuatara to Tiritiri is Oct/Nov 1999. Reasons for this are:

1) if all parties agree with the principle of the project then implementing it should take only a few months

2) the project will be more likely to be implemented if it is funded and co-ordinated by persons who have time to do so. The author has indicated that he will be available and is committed to the project for the next 12-14 months.

3) mid spring is the optimal time of the year to undertake transfer and releases of tuatara. Invertebrate food sources are most abundant during this period, and the climate of offshore islands still wet, and becoming warmer, which will encourage released tuatara to explore their release environments.
18.2. Capture of tuatara

The first visit to Ruamahua-iti to capture 60 adult tuatara is planned for mid-late October 1999. As mentioned in the proposal, the preferred make-up of the 8 personnel for this trip is:

1. University member (team leader)
2. DoC staff
3. 5 iwi representatives
4. 1 SoTM representative
5. 1 media representative

It is anticipated that at least one experienced DoC officer will accompany the team to oversee the operation. DoC may choose to fill the other position with a staff member who would benefit from training in tuatara monitoring and handling, and working in offshore island environments. A suitable candidate would be one of the DoC staff stationed on Tiritiri. It would be of immense advantage to have DoC personnel on Tiritiri trained in basic monitoring techniques for tuatara and able to identify any problems which may occur with the released animals following translocation.

Proposed characteristics of capture expedition and transport to Tiritiri:

1. Late October/ early November 1999
2. Duration of 4 days and 3 nights only
3. Transport of capture party to Ruamahua-iti by boat
4. Accommodation at the Aldermen’s Group on a boat capable of catering for the entire capture party (i.e. no camping will occur on Ruamahua-iti)
5. Transport of tuatara to mainland by this same boat once required numbers have been caught
6. Transport of tuatara by University vehicle to Gulf Harbour, Whangaparaoa Peninsula that same day and immediate transport by ferry to Tiritiri.
7. Immediate release of 48 of the 60 tuatara in their designated release sites
8. Retention of the remaining 12 tuatara (6 males and 6 females; 2 of each gender from each release site) in the modified aviary on Tiritiri until the public release day (maximum of 7 days following arrival of animals onto Tiritiri).

18.3. Preparation of release sites

18.3.1. Sites inaccessible to the public
The Cliff release site will require no preparation prior to the release as it already contains suitable numbers of natural seabird burrows to provide refuge for tuatara.

The two forest sites (Bush 1 and Bush 21) do not contain natural burrows and will require artificial burrows to be placed within the general release area. Numbers of artificial burrows should be provided at a ratio of three burrows to every two tuatara - giving a total of 30 artificial burrows to be constructed in each of these two release sites. Placement can be decided upon later, but should be based upon at least 20 of the burrows being located within the small release area itself and the reminder set in wider concentric rings around the release area.

18.3.2. Public access site

The public release site will require a similar number of artificial burrows to the forest sites. It will also require a substantial amount of site preparation to offer the opportunity for the public to view tuatara. Planning and design of structures proposed in Fig 5 will need to be incorporated into DoC annual business plans, or provision for this planning to occur dependant on the success of the releases in the other sites. Construction of facilities for both public and tuatara will need to be undertaken in winter/spring prior to the release (which at this stage is planned for 2000 at the earliest).

18.4. Preparation of holding facility on Tiritiri

A suitable holding facility, in the form of the disused aviary, already exists on the island. A minimal amount of work will be required to convert it to a secure holding facility for 12 adult tuatara. Anticipated modifications are:

- partitions to separate individual animals
- artificial burrow in each compartment
- easy access for monitoring and feeding of animals

There are issues of disease transmission in close holding facilities such as aviaries. The Tiritiri aviary has held a number of bird species in the past (temporary translocation facility), but none within the last 16 months. Issues concerning the safety of the aviary as regards disease transmission need to be discussed with DoC and Auckland Zoological Park personnel.
In 1997 six sites on Ruamahua-iti were selected for comparison of tuatara population structure between forest types and to gain an accurate estimate of the number of tuatara on the island (Fig 6). Each site was systematically searched by two persons each night for six successive nights, the first three of which were devoted to an estimate of population density (mark-release-recapture). Animals caught on these nights were individually marked at the capture location and immediately released. No other data was collected from animals, and handling was kept to a minimum. Observations were made of relative tuatara abundance over all areas of the island each night, apart from study sites (study sites were dispersed over the whole island and travelling between them covered most of the island’s area). Total area of study sites on the island was 0.7472ha (3.9% of the total island area).

Estimates of population size for a given day within study sites were calculated using the formula:

\[ N = \frac{\text{Total marked} \times (\text{total seen on that day} + 1)}{\text{(total seen on that day which were marked} + 1)} \]

Density estimates varied considerably between sites (Table 6). Additionally, some areas of the island supported very low numbers of tuatara, while other areas were estimated to support considerably greater densities than those found in the study sites. Location of higher and lower tuatara density were estimated to cover approximately the same area of ground. A very rough estimate of island population size can be calculated by separately averaging the density estimates per hectare for upper and lower estimates at each site and multiplying by the island’s area (19ha). Calculating total population size by applying density estimates over the entire area of the island can be justified on two grounds. Firstly, tuatara were found in all areas of the island and secondly, estimates of the size of Ruamahua-iti are derived from aerial photographs which do not take into account the area of viable habitat on vertical or steep sloping locations. Observations of these areas indicated that they support high densities of tuatara. Therefore, there is most probably greater than 19ha of land on the island supporting tuatara.
Table 6 Estimated population size and density of tuatara in study sites on Ruamahua-iti.

<table>
<thead>
<tr>
<th>Site</th>
<th>Area (m²)</th>
<th>number calculated/site</th>
<th>Density/ ha</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Day 1</td>
<td>Day 2</td>
<td>Day 1</td>
</tr>
<tr>
<td>1</td>
<td>1198</td>
<td>33</td>
<td>275</td>
</tr>
<tr>
<td>2</td>
<td>894.4</td>
<td>14.7</td>
<td>161</td>
</tr>
<tr>
<td>3</td>
<td>1405.8</td>
<td>18.3</td>
<td>130</td>
</tr>
<tr>
<td>4</td>
<td>1378</td>
<td>42.5</td>
<td>308</td>
</tr>
<tr>
<td>5</td>
<td>886.4</td>
<td>29.3</td>
<td>331</td>
</tr>
<tr>
<td>6</td>
<td>1710</td>
<td>126</td>
<td>735</td>
</tr>
</tbody>
</table>

Average estimates [+/− 95% C. I.] using the greatest value of each pair of site estimates gives an average density of 353 tuatara/ha [555, 152] and a total population size of 6720 [10550, 2889] animals. However, the value obtained from site 6 (day 1) varies considerably from both other site estimates and from the estimate in the same site the following day. If this value is considered to be an outlier and omitted from calculations then the average density of tuatara/ha is 277 animals [335, 220] and total population size 5271 animals [4180, 6361].

Population estimates using the lower of the estimates obtained for each site suggest an average density of 216 tuatara/ha [278, 153] and a total population size of 4098 animals [2913, 5283].
Figure 6 Location of study sites for estimating population size of tuatara on Ruamahua-iti Island.
Some papers originating from field-work during this research are included below. A range of other papers are currently being prepared from data not used for the core of this thesis. They include:

1. Red Mercury Island: translocation success & merits of captive-reared vs. wild founders;
2. Comparative efficiency of methods for sampling terrestrial invertebrates;
3. Artificial tuatara refuges and their micro-climatic similarities to seabird burrows;
4. Intra-specific competition on Moutoki Island and
5. Modelling of harvest magnitude and frequency for small populations of tuatara.

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APPENDIX ONE

1 Update On Archaeological Sites On The Aldermen Islands

Abstract:
Archaeological sites are described from Ruamahua-iti and Hongiora Islands in the Aldermen Islands group, off the east coast of the Coromandel Peninsula. These observations constitute significant amendments to previous literature, highlighting the poor human settlement information base available for such isolated islands. Factors influencing survey success and the implications for management are also discussed.

Keywords: Archaeology; Maori occupation; Aldermen Islands; Ruamahua-iti; Hongiora.

INTRODUCTION

Situated approximately 20 km off the east coast of the Coromandel Peninsula, the Aldermen Islands comprise four major islands (size range 16 - 32 ha) and at least 16 smaller stacks (Taylor 1989). All islands within the group are designated Nature Reserves and support populations of varied fauna including northern tuatara (Sphenodon punctatus) (Cree and Butler 1993), and various seabird species.

Although recognised as being used to some extent by Maori in pre-European times, the Aldermen group has attracted only superficial attention to archaeological sites. Accounts concerning occupation of individual islands have been recorded since the 1700s (see Moore 1973), but it was not until 1972 that an effort was made to document the evidence on all islands by P. R. Moore during an Auckland University Field Club expedition. However, Moore noted that the lack of time available on each island as well as other natural factors, meant that his examination represented an incomplete assessment of human occupation.

1 Current status: Ussher, G. T. In press. Update on archaeological sites on the Aldermen Islands. Tane 204
The notes presented here, together with Moore’s observations are once again viewed to represent only an incomplete picture of the true extent of sites present on these islands. It is hoped that thenotes presented here will encourage more experienced archaeologists to conduct a full inventory of the Aldermen’s pre-European history.

METHODS AND RESULTS

Observations were made during a 14 day visit to Ruamahua-iti and Hongiora Islands between November 22 and December 5 1996. Tuatara population dynamics were the focus of the expedition. As a consequence only a small amount of time was devoted to mapping archaeological sites, although on Ruamahua-iti most, if not all, of the main island area was searched and sites noted. Rudimentary maps of sites were compiled and sites measured or approximate dimensions estimated. Artefacts where found (not actively searched for) were noted and their locations mapped.

**Ruamahua-iti Island**

Seven days were spent on the island and approximately 80% of main island plateau was searched for sites. Moore notes one terraced site (U11/5), comprising a series of nine major levels, on the most northern ridge of the island (Fig. 1). On this current visit, four additional terraced sites and two artefact locations were noted.

Site U11/6: Located on the most eastern ridge overlooking Tuatara Bay (Fig. 1), this site consists of four major terrace levels, all well preserved (Fig. 2a). Maximum length of terraces is approximately 40m, and width ranges between 4m and 6m. Vegetation present is mid-successional, with kawakawa (*Macropiper excelsum*) dominating the subcanopy and understorey and *Coprosma* sp. and mahoe (*Melicytus ramiflorus*) forming a 5-6m incomplete canopy. Moderate petrel burrowing is present throughout the site, with some small areas of high burrow densities.

Site U11/7: Located on the southern-most cliff edge (running east-west) of the island (Fig. 1), this site consists of two indistinct terraces whose length is bisected by north-south running ridges ascending to the southern cliffs. Total length of each terrace is in excess of 30m, width 5m or less. Severe slumping is present in some places due to the steepness of the valley sides, with heavy burrowing below the terraces, although only minor burrowing on the terraces themselves.

Site U11/8: A major terrace system is situated along the full length of a central north-south orientated ridge (Fig. 1). The system consists of a minimum of 14 individual platforms and earthen walls, with the width of all terraces 4-6m. Length ranges from 8m (greatest altitude) to over 40m, stretching across the ridge formed between two small stream valleys (Fig. 2b). All terraces are well
preserved, especially those in the upper section of the system. Vegetation stage differs with altitude, appearing to reflect time since last modification. Upper terraces support a mixture of 7-9m karo (*Pittosporum crassifolium*) and mahoe, with an open understorey. Burrow densities are very high in these upper areas. Lower terraces are characterised by considerably denser vegetation dominated by sapling *Coprosma* sp. and mahoe. Burrow densities are significantly lower or absent from these lower areas.

Site U11/9: On the next ridge west of U11/8 is single, well preserved terrace, 15m long.
Site U11/10: On the next ridge east of U11/8 is a single 20-25m long, 4m wide incomplete terrace. At the western end of the terrace an earthen wall runs diagonally towards the west, until the terrace peters out. It appears that the terrace was at an advanced stage of completion, with earth being gradually dug out from the bank.

Artefact Find Sites: Two rounded, smooth oblong stones were found on the banks of streams on the island (see Fig. 2). Both stones showed obvious signs of being deliberately crafted by humans. However, since neither artefact was removed from the island for examination, their specific use is not known. Small amounts of obsidian (all flakes) were also found throughout the island (not mapped).

**Hongiora Island**

Seven days were also spent on Hongiora. However, the extreme burrow densities over the entire island meant that only two small sections (approximately 10% of the island area) on the southern and north-eastern coasts were searched. On the last day a brief search was also made on the southern peninsula, yielding a significant defensive formation. Moore's 1972 visit (of several hours) found no archaeological sites on Hongiora but noted two obsidian fragments on the south-eastern side of the island. In contrast to Ruamahua-iti, much of Hongiora is covered in thick, low taupata (*Coprosma repens*)/ hymenanthera (*Hymenanthera novae-zelandiae*) vegetation which makes observation of local topography difficult.

Sites U11/11 & U11/13: Located at the coastal fringe of the single large south-facing valley on the island, this site consists of three separate piles of heaped stones, one near the coastal edge and the other two on the side of the gently sloping valley on the small ridge to the west of the valley (Fig. 3).

Site U11/12: This was most probably a defensive paa site. Located on the small peninsula at the south end of the island (Fig. 2), there has been considerable earth and rock works over the entire peninsula (Fig. 4). The peninsula is accessible only by a 3-5m wide tongue of land to the main
island, with 15-20m cliffs on either side of the tongue and surrounding the entire peninsula. The top of the area has been worked to provide a flat area approximately 30x 15m. A 2-3m rock retaining wall above the southern and western cliffs provide support for backfilled land facing the mainland. On the eastern side, less steep banks have been modified to provide four terraces stretching the length of the peninsula, with a maximum length of 30m. Terraces at the western end of the peninsula are in an advanced state of decay, while those at the eastern end remain well preserved. Due to lack of time, no searches were conducted for artefacts on this site.

Artefacts: At least ten obsidian fragments were noted, most occurring at the bottom of the southern valley.

DISCUSSION

This survey highlights important problems inherent in island studies. Although access to sites and available survey time influence mainland site assessments, the quality of island appraisals is often significantly influenced by the additional problems of vegetation cover and density of seabird burrows.

Time and vegetation stage were undoubtedly the leading factors accounting for the differences between this and Moore's 1972 survey. Although Field Club members spent 10 days on Ruamahua-iti in 1972, identification of archaeological sites was secondary to the noting geological features on the island, meaning that only a minimal amount of time was spent on the island proper, and then mostly within valleys due to vegetation thickness elsewhere (Hayward, pers comm.; Court et al. 1973). Considerable ridge terracing was immediately apparent on this recent visit to Ruamahua-iti, but the maturity of the vegetation since Moore’s survey 27 years ago means that topographical features were no longer obscured to the extent that they would have been in 1972.

From this survey it is clear that the two islands visited, and probably also others in the Aldermen Islands group, have been modified considerably more than suggested by Moore’s 1972 findings. Evidence of pre-European landscaping covers all three of the major internal ridges and all three major valley heads or sides ending in cliffs on Ruamahua-iti Island. This suggests that most of Ruamahua-iti’s typically north facing ridges and valley sides, perfectly positioned to receive maximum radiation, were managed as gardens. Similarly, evidence of occupation found in the small areas searched on Hongiora suggest that this island too underwent considerable vegetation removal, both for gardens and for defensive areas. The piles of stones located at the periphery of the main southern sloping valley suggest that the area was intensively managed as gardens, for
which the clearing of rocks is a pre-requisite. The lack of defined terracing on the areas examined on Hongiora can be attributed to the gently sloping nature of the valley, the dense undercover present and, more importantly, to the extreme degree to petrel burrowing which would have quickly broken down any evidence of small earth walls or other structures. Hongiora is characterised by its high petrel densities, and this gives the island a soil quality unmatched by other islands in the group. Bell et al. (1951) noted this fact when documenting the quick recovery of vegetation on the north face of the island following a fire in 1935.

Islands offer one of the best localities for documenting and preserving not only our biological, but also our historical heritage, both Maori and European. Difficulty for human access and enforced landing restrictions have ensured that sites on many islands such as the Aldermens largely suffer only natural biological damage (such as from burrowing petrels), rather than that caused by heavy human traffic and associated activities. Allocation of protection effort depends on our ability to adequately record the type, distribution and quality of sites in areas. Failure to document significant finds could mean lost anthropological opportunities and, in the case of poorly understood areas such as the Aldermens, could heavily influence how human occupation is interpreted in terms of habitat degradation and present patterns of species distribution.

The Aldermen Islands fall into an elite category of sites. Many islands, although featuring sites of significant value have also suffered significant European habitation, which in many cases has destroyed, modified or confused the extent of Maori occupational evidence. This features heavily not only on many inshore islands (such as Tiritiri Matangi, Mana and Motutapu Islands), but also on many offshore islands considered to have escaped such affects - for example Red Mercury Island (Mercury Group), and two of the three Chickens Islands in the Hen and Chickens Group. Additional value is added by their current high wildlife protection status restricting research impact and the islands' unique characteristics (high transport costs, isolation, landing difficulties, and lack of camp sites and facilities) restricting research interest.

The pre-European history of the Aldermens is poorly documented. This study, again limited by time and lack of experienced surveyors, at least doubles the available information base. Conditions for conducting surveys on some islands will continue to increase favourably as vegetation matures. However, site quality will continue to degrade as petrel burrowing continues and future research and management projects ensure an ongoing human presence. Any attempts to catalogue past uses for these islands must consider the likely damage surveyors may make to petrel burrows, which in many places frustrates or totally prevents. Nonetheless, considering the favourable survey conditions and ongoing threats to sites, it would seem timely for a new, concerted effort to document the human history of these islands.
ACKNOWLEDGEMENTS

I am extremely grateful to David Mawer, (Isles of Scilly, England), for help in the field and for the enthusiasm he showed in discovering the Aldermen Islands' history during our stay. The funding for this survey was generously provided by the Waikato Branch of the Royal Forest and Bird Protection Society (NZ) and Auckland University. Many thanks also to John Craig (Auckland University), Bruce Hayward (Auckland Museum) and Neville Richie (Waikato Department of Conservation) for commenting on drafts of the manuscript.

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Figure 1. Ruamahua-iti Island. Size 25 ha. Most of main island area was searched apart from southern slopes below ridge-line and eastern slopes below ridge-line.
Figure 2. (a) Site U11/6 showing four distinct terraces (T1-T4) on the eastern cliffs ridge and (b) Site U11/8 showing 14 discrete terraces (T1-T14) and earthen walls.
Figure 3. Hongiora Island. Size 16 ha. Approximately 10% of the main island area was searched for archaeological sites, concentrated mainly around the southern landing and north-eastern valley areas.
Figure 4. Site plan for fortified peninsula, (U11/12) on Hongiora Island showing high degree of earth and stone-works.
Cost-efficient Pit-Fall Traps For Invertebrate Sampling

Abstract
A cheap and effective pitfall trap design is described. Constructed from a single 2L 'softdrink' bottle, this trap has been extensively field tested and offers a light-weight trap which meets all requirements for pitfall trapping with a minimal financial outlay.

Keywords: pitfall trap, softdrink bottle, invertebrates.

INTRODUCTION

Pit-fall traps offer one of the fastest, most inexpensive methods of sampling invertebrate faunas. Used as specified (see Southwood 1978, Adis 1979, and Topping & Sunderland 1992 and refs therein for assumptions and critical variables), pitfall traps yield large data returns readily available to statistical analyses, with minimal effort and maintenance.

New Zealand studies have, in the past, used a variety of trapping. Many designs are elaborate, using a variety of external and internal sleeves and collection vessels (e.g. Moeed and Meads 1985) increasing material and labour costs and weight of prepared trap. The trap proposed by Moeed and Meads is the most widespread alternative design used in New Zealand studies.

The following notes detail a light-weight, inexpensive pit-fall trapping design with low labour and maintenance costs, is quickly installed, and provides sufficient dimensions to ensure efficient sampling of broad ground-based invertebrate taxa.

TRAP DESCRIPTION

Design

The core unit consists of a single 2L 'DYC' brand or softdrink bottle. DYC bottles, used to store vinegar, are preferable to normal softdrink containers because of their non-tapering sides and indented rings for added structural strength down their length. To construct the pitfall follow the steps detailed below:

1. Remove the head and neck of the bottle by cutting at the point where the neck meets the sides (Fig. 1). The external diameter of the pitfall will be approximately 110 mm.
2. Invert the top and cut out the head to provide an open funnel.
3. Bury the bottle body upright and level with ground surface, cut drainage holes in the bottom.
4. Place the collection cup (empty for live collection, preservative filled for kill-trapping).
5. Insert the funnel until the bottom meets the cup rim, ensure the sides of funnel are flush with the body wall.
6. Cover with metal or wooden rain shields. Shields are held in position by four corner 'legs' either constructed as part of the main cover body (metal shields), or attached afterwards (wooden shields).

Field Testing

The 2L pitfall has been in use on a number of New Zealand offshore islands over the last 5 years. Pitfall traps on the Chicken Islands, Northland have been on site for over 4 years continuously (although set for only short periods during this time), and various other locations for at least 1 year. Original designs produced problems with bottle deformation after 2-3 weeks following installation, presumably due to differing pressures inside and outside the main sleeve. This problem was easily solved by providing drainage holes in the bottom of the external sleeve. These holes are also necessary to drain excess fluid which may overflow into the external sleeve during heavy rainfall. In total, 100 2L traps have been or are in use by the author and 45 in a 6 month study by another graduate student. In addition, 50 2L traps are in use by the Department of Conservation in studies monitoring changes in invertebrate populations on various Hauraki Gulf Islands (Chris Green, pers comm.).

Catch profiles indicate that the trap is capable of sampling invertebrates within the range of animal sizes available in the environment - as would be required for such as trap. Upper catch sizes experienced include Arachnids up to 40 mm total length, beetles up to 30 mm, Orthopterans up to 40 mm, centipedes 100 mm and skinks up to 100 mm body length.
Cost and Availability

2L softdrink bottles are one of the most plentiful resources around. Used bottles are available either free or for a minimal fee from recycling centres. New, unused bottles, are readily available from the manufacturer (usually free) from their reject or quality control lines during production. Plastic collection cups come in a wide variety of materials and sizes. 300ml plastic party drink tumblers provide a sturdy cup with sufficient volume to hold preservative when trapping continuously for a number of weeks or months. The cost of cups varies, but works out at approximately 15 cents per unit. Overall, an entire pitfall unit (not including cover) can be assembled for as little as 15 cents, and takes around 10 minutes to make.

ACKNOWLEDGEMENTS

The tuatara translocation projects in which the author is using this pitfall design are supported by WWF (NZ), NZ Lotteries Grants Board, Auckland University, the Waikato branch of the Royal Forest and Bird Protection Society (NZ), and Garmont Footwear Industries. Many thanks to Chris Green (Department of Conservation) and John Craig (Auckland University) for commenting on drafts of the manuscript.

REFERENCES


Figure 1. Materials and installation of the '2L' pitfall trap.
Method for Attaching Telemeters to Medium-sized Lizards: Trials on Tuatara (Sphenodon punctatus)

INTRODUCTION

The need to know why species translocations succeed or fail places added emphasis on post-release monitoring of individual animals. In New Zealand, a rare endemic reptile, the tuatara (Sphenodon punctatus), has recently been re-introduced to the first of many locations within their historical range. Tuatara are nocturnal, extremely cryptic, and often inhabit deep underground burrows which make them difficult to monitor. Hand catching of tuatara when they are active above-ground has traditionally been the most efficient method of gaining morphological or dispersal information from individual animals, but can result in considerable environmental disturbance and disruption of tuatara natural activity patterns (see Cassey and Ussher 1999 for alternatives). Investigations planned for the use of habitats by translocated tuatara required that a more efficient and less invasive method of yielding information be developed. Miniaturised radio telemeters (transmitters) were viewed as the most favourable method of obtaining this information.

Tuatara and radio transmitters

The history of transmitter use with tuatara is brief. Hind-quarter mounted transmitters and temperature probes were used in the late 1960's-1970's, although attachment was for periods of 9 days or less (Barwick 1982). Tracking trials during 1976-1977 by New Zealand government scientists used a leather harness system for the then bulky transmitters (Newman 1980), but the design was not adopted because of fears for tuatara safety. Direct fixture of the transmitter to the dermis using

surgical tape (C. Tyrrell pers. comm.) or with adhesive glues (B. Goetz. pers. comm.) has been trialled, although duration of trials on captive tuatara have been less than 4 weeks and adhesion beyond 9 days in the field has yet to be successful.

The translocation study undertaken by the author required that the attachment method be effective for periods of 4 months or more (long-term attachment) because of an inability to visit offshore island study sites on a regular or predictable basis. Potential avenues for long-term transmitter attachment were also constrained by aspects of tuatara biology. Tuatara grow to 700-1000 mm in total length and weigh from 300-800 gms, meaning that force of contact with ground surfaces, and therefore wear on harness materials during locomotion, can be considerable. Compounding this is the tendency for tuatara to use small crevices as retreats, often amongst abrasive rock piles. Additional problems are an annual moult cycle, susceptibility to dermal infection under non-porous skin coverings and extremely slow rates of healing following surgical procedures (such as sub-dermal telemeter implantation) (R. Jacob-Hoff pers. comm.).

Trials on captive animals by the author focused on indirect transmitter attachment at the anterior region of the animal, using a modified harness system similar to that employed by Fisher and Muth (1995) for small horned lizards. The design accepted for use was a modification of the harness constructed by N. Nelson (Victoria University, New Zealand) and D. Newman (Department of Conservation, New Zealand). This design was successfully trialed for 2.5 months (the average period between sampling visits to the islands) on captive animals and met additional criteria of being simple in design, readily repairable, and able to be constructed in the field from inexpensive materials. Below, I detail the final design of the transmitter harness which can also be used on other similar-sized reptiles.

Radio transmitter and harness package

The radio transmitters used were tear-drop single stage transmitters (Sirtrack Ltd) measuring 20 mm x 10 mm with a 70mm whip antenna and weighed < 4 g (Fig. 1). Ventral surfaces of transmitters were flat to maximise contact with transmitter pad and aid in fit to tuatara body shape. The harness comprises two major sections: the transmitter pad, and the backpack straps (Fig. 1). The pad is constructed from a 30 mm x 20 mm strip of porous polypropylene webbing. Edges are heat sealed and smoothed to prevent abrasion of the dermis. Two lengths of polyester 13 mm wide braid elastic are sewn to each side of the longer edge of the pad. Two thin retainer straps of 4mm wide polyester cord are sewn across the pad to act as anchors for the transmitter sitting atop the pad. Cyanoacrylic glue (Superglue™) is used to affix the transmitter to pad. When fitting the harness, the transmitter pad is placed on the upper ventral surface directly above either scapula with the transmitter aerial pointing
posteriorly and the two straps pointing dorsally up and over the back. Pull the most anterior strap dorsally over the back, down anterior to the forelimb, under and across the ventral surface to rise posteriorly to the other forelimb and meet the diagonally opposite edge of the transmitter pad.

Repeat the procedure with the other strap, passing it over the back, down posterior to the forelimb and diagonally across the ventral surface. Pull both straps firmly to the pad edge, and measure for trimming and attachment. Ensure fit is tight enough to account for the probable thorax expansion being exerted by the animal (adpression of the ribs when the animal relaxes following harness attachment will reduce tension in harness straps). Trim straps to the required length and heat seal the ends to prevent fraying. Attach straps to the pad with cotton thread using no more than 4-5 simple stitches. This will serve as a 'weak link' which will part if the animal becomes entangled and will release it from the harness (and transmitter). Final weight of the transmitter package (transmitter + harness) is < 6 g.

**Evaluation of harness package**

For this study, 29 tuatara were fitted with backpack transmitters for a total of 8,300 days. Harness performance was assessed by looking at wear of materials, reliability of weak links and gross impact on animal welfare and behaviour. Wear on materials over the 14-16 months was negligible in most cases. Harnesses endured temperatures from 3°C (winter) to 25°C (summer), frequent wetting and drying, abrasion against rock surfaces, and inundation with particulate soils and mud. Harness straps maintained elasticity throughout the study and webbing pads did not deteriorate. A single elastic strap frayed slightly at the edge. The weak links exceeded performance expectations. Trials on captive animals identified minimum stitching required to part under strain exerted by different sized tuatara (this may differ for smaller or larger lizards). Unfortunately, environmental degradation of weak links was not sufficiently taken into consideration. Harnesses worn by animals throughout the spring and summer months (minimum duration of 120 days worn) were discarded when autumn rains deteriorated the exposed cotton link fibres.

Impact on animal welfare was determined by presence of harness-induced injury to individuals or as loss of body condition compared to tuatara without telemeter packages released at the same location. Animals in one release population (n = 9) showed no signs of physical injury, but 80% (n = 20) of tuatara on the other island exhibited varying degrees of abrasion under one or both axillae. This effect was attributed to the larger-sized tuatara on the unaffected island (mean snout-vent length (SVL) males: 265 mm (± S. E. 10 mm); females 247 mm (± S. E. 23 mm), while those on the affected island were inherently smaller individuals (SVL males: 203 mm (± S. E. 4 mm); females 174 mm (± S.
E. 1 mm). This influence of body size on harness affect did not eventuate during trials, probably because tuatara in trials were large animals of similar size to those on the unaffected island.

In most cases (13 out of 16), abrasion only resulted in damage or removal of scales within a small area of the axilla. In 3 cases, abrasion was sufficient to also break the skin surface, but abrasions healed within 2 months. It is probable that, shortly after fitting, the elastic straps of the harness stretch a small amount and conform to the animal's body shape, with most abrasive stress being applied during this easement period. Replacement harnesses were fitted less firmly to animals and no further problems of strap-induced abrasion were recorded. Additional abrasive problems in the form of small cuts to the dorsal spine projections were seen on three animals, and seem to have also been caused by overly tight harnesses.

The impact of harnesses on foraging and mating behaviour was of paramount concern at the study's conception. Increased probability of predation was not an important consideration, as it is unlikely that the natural predators of tuatara are able to prey on adult individuals in forested habitats (but see Saint Girons et al. 1980 for predation by Australasian harriers (Circus approximans) in modified grassland habitat). All individuals fitted with transmitter packages were mature adults. Analysis of faecal pellets from transmitted and non-transmitted animals indicated no recognisable difference in either the type or size of prey appearing in diets. Effect of transmitter packages on foraging efficiency would manifest itself as long-term weight loss in affected animals. Weight changes for tuatara fitted with transmitters were comparable to weight changes for animals without transmitters (e.g. release population 1: % body weight change since release for tuatara with harnesses = -3.6% - 0.9% (n = 2); without harnesses = -1% - 10.8% (n = 5) over the same period). Additionally, mating behaviour was exhibited by both transmitted and non-transmitted animals during the study. Differential mating success of transmitted and un-transmitted animals was not followed.

In summary, the backpack harness system for attaching radio transmitters to medium-sized reptiles such as tuatara is a safe, reliable design which ensures the safety of the study animal and enables long-term (> 4-5 months) attachment. Correct fitting of harnesses will prevent abrasion seen on some tuatara in this study. Materials used for the harnesses weathered New Zealand seasonal climatic variations as well as the environmental degradation caused by tuatara habits. Weak links incorporated to protect study animals need to be improved to ensure that harnesses are not shed solely due to climate-induced degradation of cotton links. Perhaps sealing cotton links in weatherproof materials will ensure greater efficiency. Harness materials may also have to be modified for use on smaller lizards to prevent abrasion-related problems seen in this study.

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LITERATURE CITED


