# Conservation ecology of kea (*Nestor notabilis*)

WWF
NEW ZEALAND

1 August 1999

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# Executive Summary

## Conservation ecology of kea (*Nestor notabilis*)

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## Investigation overview

Kea (*Nestor notabilis*) are large green parrots endemic to the montane areas of New Zealand's South Island.  An extensive amount of their habitat is reserved in the New Zealand system of national parks, within which kea are currently widespread (Bull *et al*. 1985), albeit at a very low density of approximately one bird per square mile (Jackson 1960; Clarke 1970; Bond & Diamond 1992).

Kea are presently classified by the Department of Conservation as a second priority threatened species (Molloy & Davis 1994), but remarkably little is known about the size and status of the wild population (Diamond & Bond 1999; Temple 1996).  Anderson (1986) guessed a total population size of 1000-5000 individuals by extrapolating a density of one bird per square mile to the rest of the Southern Alps. A more precise estimate is not available due to the lack of a reliable, cost-effective survey method.

A total kea population of 5000 (the upper limit of Anderson's guess) is small enough to conclude that the kea population has declined to some extent since the late 1800s – mid 1900s.  During this period, kea abundance was such that 150 000 individuals were legally killed under a government bounty scheme.  To enable such a cull to take place, kea must have been considerably more abundant than they are today.   The anecdotes of old-time high country users support this conjecture; encounters with large flocks of kea are now relatively rare.

There are two possible explanations for this apparent decline. The first of these is that kea numbers were artificially elevated in the late 1800s – mid 1900s by a superabundance of food in the mountains. This food came in the form of sheep and deer flesh which kea learned to exploit. Prior to European settlement, kea numbers were apparently also quite low, as kea were relatively inconspicuous to both Maori and early European settlers (Diamond & Bond 1999). The low kea abundance of today may simply be a return to a state that is limited by the natural food supply in montane areas, as sheep grazing in kea habitat is now rare and deer numbers have been greatly reduced by government funded control operations. If this explanation is true, then the current low density of kea is probably no cause for alarm.

A second explanation for kea being rare today is that predation and/or competition from introduced mammals has had a detrimental effect on kea population dynamics. Several species of New Zealand birds living within intact forest reserves are threatened (or have been driven to extinction) by introduced mammals (Clout & Saunders 1995; Holdaway 1989; Innes & Hay 1991; McLennan *et al.* 1996; O'Donnell 1996). If predation and competition continue to impact on kea numbers today, then kea numbers are likely to continue to decline unless some form of wildlife management is initiated to reverse this trend.

This study aims to investigate whether kea numbers are in a continuing decline. Predation was chosen as a research priority because it has the potential to cause a very rapid decline in kea abundance. Also, predation at the nest has been strongly implicated in the declines and extinctions of several other species of cavity-nesting birds (O'Donnell 1996). For example, in a recent study of kaka (*Nestor meridionalis*; a sympatric congener of kea), an alarming rate of nest predation and concurrent mortality of adult females occurred. Out of 20 kaka nests, 9 young kaka and 4 adult females were killed on the nest, with only 4 chicks surviving to independence (Wilson *et al.* 1998).

This study investigates the proportion of kea nests that fail and the proportion of adult females that survive nest failure. We also investigate the frequency of death among adult

kea in the course of their activities away from nests and suggest priorities for future research into the conservation ecology of kea.

Throughout this report, the term 'hinterlands' is used to describe areas in which kea feed only on naturally occurring food (i.e. where there are no skifields or refuse dumps).

## Objectives

- Determine the proportion of kea nests that fail
- Determine the proportion of nesting females that are killed when nests fail
- Determine the annual proportion of adults that die/are killed during their daily activities away from nests
- Measure productivity among kea who have access to food from a skifield carpark for three months of the year
- Identify the demographic parameters upon which a study of kea in the hinterlands should focus
- Develop and begin to implement a strategy for studying the population biology of kea in the hinterlands

## Peer review and publication of this report

This report consists of material contained within an unpublished MSc thesis prepared by Josh Kemp under the supervision of Dr Ian Jamieson (Senior Lecturer, Department of Zoology, University of Otago) and Dr Graeme Elliott (WWF-NZ Project Officer; Senior Scientist, National Kakapo Team). The results were presented at the 1999 Conference of the Captive Management Advisory Group (CMAG) and the 1999 Conference of the New Zealand Ecological Society.

The Department of Conservation (St Arnaud Area Office) has considered the recommendations contained within this report and have secured funding for a 1999/2000 field season. We will submit our results to appropriate scientific journals when data from this season is available.

# Study site and methods

- The study area is centred upon Rainbow Ski Area in the St Arnaud Range, Nelson Lakes National Park. The surrounding valleys support intact Southern Beech (*Nothofagus* spp.) forest with alpine grasslands above the treeline.

- Adult male kea are drawn to the skifield from the surrounding valleys to scavenge for food in the carpark. Here they were captured and fitted with radio transmitters, which enabled us to locate nests and monitor survivorship.

- From 1993 to 1998, an intensive radio-tracking effort aimed at finding nests was undertaken in September-October, and any nests found were subsequently checked every two-three weeks. The proportion of nests that fail was calculated from daily nest survival rates because some nests were under observation for only part of the nesting period.

- To monitor survival, signals from radio-tagged adults were obtained at least weekly and the birds were sighted 5-20 times during each spring. If a signal failed to fluctuate and came from the same direction for several days (suggesting a lack of movement) the bird was visited on foot to see if it had died.

- In 1997 and 1998, fully-grown nestlings were radio-tagged with 5-year transmitters to monitor their survival and establish a sample of study birds in the hinterlands.

- These juveniles were located from a radio-equipped fixed-wing aircraft (Cessna 172) in January 1999. Each bird was then visited on foot to see if it was alive.

# Results

- 36 kea nests were monitored between 1993 and 1998. Breeding was widespread in the study area in all years except 1996 when we could not find any nests.

- Eggs or chicks disappeared from 31% of the 36 nests, 57% fledged at least one chick, 6% failed due to natural accidents at the nest site, and 6% were not monitored to their final outcome.

- From daily survival rates, the proportion of nests fledging at least one chick was 51% (95% CI = 34-73%).

- The probability of nest contents (i.e. eggs or chicks) disappearing decreased with increasing slope at the nest site.

- Nest predation did not increase in the breeding season following full beech mast-seeding in 1995, when stoat (*Mustela ermina*) numbers were beginning to rise.

- 37 different adult kea (8 females and 29 males) were radio-tagged. Most of these were monitored for more than one year, giving a total sample of 89 kea-years (19 female and 70 male).

- Five deaths of radio-tagged adults occurred, giving an adult survival rate of 93% (SD = 2.8%). All five deaths were of adult males.

- Adult females had to be captured at their nests because they seldom visit the skifield. This was time consuming and we sometimes had difficulty recapturing these females when their transmitter batteries went flat after two years (transmitters with 5-year batteries only became available toward the end of the study). We therefore settled with colour-banding females rather than radio-tagging them.

- Five adult females had been colour-banded prior to their nest contents disappearing. All five were confirmed alive after their nest contents had disappeared.

- Uncertainty about the fates of the other 6 adult females whose nest contents disappeared led to an estimated annual adult survival rate of 82%.

- Productivity around the skifield was 0.33 female chicks per adult female per year.

- Population modeling showed that a change in adult survival has roughly four times the impact on population growth as a proportional change in any other parameter.

- 11 fully-grown nestlings were radio-tagged in 1997 and 7 were radio-tagged in 1998. A third cohort of 9 will be radio-tagged in1999.

- Of the 11 nestlings radio-tagged in 1997, 10 signals were obtained during the tracking flight a year later. Follow-up tracking on the ground showed that 8 of the nestlings had survived their first year, 1 had died, and one had dropped its transmitter. This yields a fledgling survival rate for 1998 of 89% (95% CI = 70 – 100%).

- The radio-tagged one-year-olds led us to four flocks of kea which contained unmarked one-year olds, indicating that breeding was probably widespread among kea in the hinterlands in 1997.

## Conclusions

Kea nests appear to be relatively immune to predation from introduced mammals. The nest survival rate was significantly higher than that observed among kaka nests in an adjacent forest, where only 2 of 20 nests found over a period of 11 years fledged chicks (Wilson *et al.* 1998). A possible explanation for the relationship between slope and nest failure is that predators tend to avoid very steep sites. This could occur because predators probably never learn to hunt in very steep places in beech forest because few prey animals are found there. Kea nests occurred at a very low density (one nest per 4.4 km$^2$) and other stoat prey (rodents and large invertebrates) are probably more abundant on gentler slopes. Nesting in spring may also afford kea some protection from predation by stoats, particularly following beech mast seeding, as stoat numbers are normally low until January-February. With the data at hand we can only speculate about the reasons for the low predation rate. Our results agree with a previous study of kea nesting at Arthur's Pass, where no evidence of significant nest predation was found (Jackson 1963). Since most kea habitat is similar to that found within Nelson Lakes and Arthur's Pass, it seems likely that kea nesting success rates and adult survival rates are high throughout their range.

The dynamics of our study population are driven largely by adult survival. This is typical of populations of long-lived animals with low-reproductive output (K-selected species). A decrease in the adult survival rate will cause a much more rapid decline than a proportionally similar decrease in any other parameter. A rapid decline in kea numbers would, therefore, most likely be caused by inflated adult mortality. We found that survival of radio-tagged adults was high (93%) and adult females seemed to escape being killed during the few nest predation events that we observed. Since adult survival was high and a reasonable number of chicks were fledged, this population is unlikely to be

declining rapidly. The sample of colour-banded adult females whose nests failed was small, however, so we are cautious about placing too much weight on this result until the recommended further research is complete (see below).

Competition for food from browsing mammals also could cause a decline in kea abundance within national parks. Given the high adult survival rate found in this study, competition would most likely impact on kea abundance through reducing productivity and/or reducing juvenile survivorship. The development of radio-transmitters with five years of battery life makes it possible to study these parameters. A research strategy involving the radio-tagging and monitoring of three fledgling cohorts was developed during the course of the predation study and this research is ongoing.

## Recommendations

Given the results presented in this report, we make the following recommendations-

1) The locations of 29 kea nest holes are known. As nest holes are often re-used, this provides a unique opportunity to quickly find nests without the use of radio-telemetry. We recommend that each of the 29 nest holes be visited yearly in September. If they contain nests, they should be re-visited in December to see if they have been successful. At the very least, this should happen in the breeding seasons following heavy beech seeding when stoat numbers tend to increase.

2) An attempt should be made to colour-band or radio-tag the adult females associated with these nests so that their fate after nest failure can be determined. The number of colour-banded adult females in the study area stood at seven at the end of the 98/99 field season and the number of radio-tagged adult females stood at one.

3) The fledgling cohorts described above should be located and visited each summer for the next five years to improve knowledge of age-specific mortality and dispersal. Ideas for a survey method may come to hand in this process.

4) When these cohorts mature, radio-transmitters should be replaced on the females and fitted to the mates of the males to produce a sample of 20 radio-tagged adult females

in the hinterlands. Adult female survival and productivity in the hinterlands can then be estimated from these birds. We anticipate that this will involve 8-12 person-weeks of work per year, at a cost of less than $10 000 per year.

This may be a rare case where future research is desirable not because the population appears to be in decline, but because the conservation status of kea could potentially be downgraded with confidence if the necessary data are obtained.

# Acknowledgements

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# Table of Contents

# List of figures

## Chapter 2

## Chapter 3

# List of tables

## Chapter 2

## Chapter 3

# 1. General Introduction

Until now, research into kea population biology has generally taken place at high-country refuse dumps and skifield carparks where kea congregate to scavenge food from humans. These studies have relied on observations of kea marked with leg-bands (Bond & Diamond 1992; Clarke 1970). This approach has provided only very rough estimates of population parameters because of an inability to distinguish between dispersal and mortality when banded birds disappear. This is a particular problem given the great mobility of kea, in particularly of pre-adult birds (Wilson 1990). Even if there was evidence of stability in numbers of kea around rubbish dumps and skifield carparks (such as suggested by Bond & Diamond 1992), this may not be representative of population processes in the hinterlands. This is because kea might preferentially settle near these sites to have access to supplementary food. The result of these considerations, combined with the lack of a reliable, cost-effective method with which to survey kea numbers in the hinterlands, is that a great deal of uncertainty exists about the conservation status of kea.

The main reason for concern about the conservation status of kea is that kea numbers today appear to be much lower than they were in the late 1800s – mid 1900s. During this period, kea abundance was such that 150 000 individuals were legally killed under a government bounty scheme. To enable such a cull to take place, kea must have been considerably more abundant than they are today. The anecdotes of old-time high country users support this conjecture; encounters with large flocks of kea are now relatively rare (see Diamond and Bond 1999 and Temple 1996 for review). This is despite the fact that an extensive amount of their habitat is reserved in national parks. There are two possible explanations for this apparent decline. The first of these is that predation and or competition from introduced mammals has reduced productivity and increased adult mortality such that recruitment is insufficient to balance mortality. Several species of New Zealand birds living within intact forest reserves are threatened (or have been driven extinct) by predation and competition from introduced mammals (Clout & Saunders 1995; Holdaway 1989; Innes & Hay 1991; McLennan *et al.* 1996; O'Donnell 1996). There is strong evidence that these declines have been largely due to

predation upon nests, a mechanism which has a particularly strong impact on cavity-nesting species, because nesting adult females are frequently killed (Elliott 1996a; Wilson *et al.* 1998).   Kea might also be prone to predation, especially given that their nest cavities are situated on the ground.  If predation and competition are a major cause for low kea numbers found today, then kea numbers are likely to continue to decline into the future unless some form of wildlife management is initiated to reverse this trend.

A second explanation for kea being rare today relative to fifty years ago is that kea numbers were artificially elevated from the late 1800s until the mid-1900s by a superabundance of food from the presence of sheep and deer in the mountains.  Jackson (1960) observed that kea would find a deer carcass within a few days of it's being shot, and then congregate to feed on it for several days until it was finished, indicating that these mammals may have been an important food source.  Over the last fifty years, fewer sheep have been grazed in the alpine zone and deer numbers have been greatly reduced by government funded control operations.  Kea numbers may have declined as a result of this reduction in their food supply.   Prior to European settlement, kea numbers were apparently also quite low, as kea were relatively inconspicuous to both Maori and early European settlers (Diamond & Bond 1999).  The kea abundance of today may simply be a return to a state that is limited by the natural food supply in montane areas.   If this explanation is true, then the current low density of kea is probably no cause for alarm, as long as recruitment is sufficient to balance mortality.

This study seeks to determine whether kea numbers are in an ongoing decline.  The development of radio-telemetry tools enabled us to investigate individual aspects of kea demography such as survivorship and productivity.  This was our only option given the absence of a suitable method with which to survey abundance.  We hoped that in the process we might develop a new survey method for use in the hinterlands.  Even with radio-telemetry as a tool, we anticipated great logistic difficulty in a study of kea population dynamics, so we began with a study area in which most juveniles and adult males scavenge food from a skifield carpark for three months of the year (August – October).  This enabled us to quickly develop radio-tracking techniques and rapidly

answer questions about nest predation. We believe that the skifield would not exert an influence on the rate of nest predation. Other parameters estimated in the process (such as the number of fledglings produced per successful nest) should qualitatively reflect vital rates in pre-historic times, making them useful for constructing models of population growth with which to perform sensitivity analyses. Sensitivity analyses can then be used to guide future research into the conservation ecology of kea in the hinterlands (Hamilton & Moller 1995).

The remainder of this report is structured as follows-

- Chapter 2 investigates kea nest survival and addresses the issue of nest predation as a threat to the continued viability of the kea population.
- Chapter 3 presents estimates of other population parameters and integrates them into an analysis of kea population dynamics. This analysis is designed to indicate the likelihood that kea numbers are in rapid decline and to provide guidance for future research aimed at increasing confidence in this diagnosis.
- Chapter 4 summarises the research findings and presents a strategy for future research into the conservation ecology of kea.

Chapters 2 and 3 have been prepared in the form of papers for eventual submission to scientific journals. Some repetition of material has therefore been inevitable. Some readers may be concerned about the amount of inference and conjecture that is drawn from small data sets. The current climate in conservation funding, however, combined with current trends in biodiversity loss within New Zealand, dictated that this be so. Skeptical readers should bear in mind the length of time over which data was collected (6 years) and the relative simplicity of the ecosystem (*Nothofagus* forest) in which most kea live.

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# 2. Nesting success of kea (*Nestor notabilis*) in New Zealand Southern Beech forest.

## Abstract

To gauge the extent to which predators compromise kea (*Nestor notabilis*) nesting success, 36 kea nests were observed in Southern Beech (*Nothofagus* spp.) forest in Nelson Lakes National Park from 1993 to 1998. Egg laying occurred from late-July to mid-September and chicks fledged by mid January, excepting two late nests in 1998. The survival rate of nests was high (51%) compared to 10% for kaka (*Nestor meridionalis*) in an adjacent Southern Beech (Wilson *et al.* 1998) forest. The probability of nest failure significantly decreased with increasing slope of the nest site. Nest predation did not increase in 1995 following full beech mast seeding when stoat (*Mustela ermina*) numbers were rising, but most nests had fledged by the time stoat numbers presumably reached 'plague' levels in late summer. The relatively low nest predation rate is likely to result from 1) kea often nesting at high altitude and in steep places where predators tend not to find them (they probably never learned to search in these places because the density of kea nests was very low), and 2) kea breeding in early spring (before stoat numbers rise in January). There was no evidence of predators killing adult females at nests, but the sample size was small. These results support those of a previous study (Jackson 1963) where no significant nest predation was noted.

## Introduction

Kea (*Nestor notabilis*) are large (males = 930g, females = 840g) green parrots endemic to the South Island of New Zealand. As kea typically inhabit mountainous areas of the Southern Alps, most of their habitat consists of steep slopes dominated by Southern Beech (*Nothofagus* spp.) forests and alpine grasslands. While they often forage and socialise in open areas above the treeline, kea virtually always nest within the forest, using natural cavities on the ground (Jackson 1963).

Kea are one of seven species of indigenous cavity nesting birds that were previously common in South Island beech forests (Table 1). Two of these are now extinct on the mainland and the ranges of four others have contracted to small strongholds. Predation at the nest by introduced mammals has played a key role in these declines and extinctions, many of which have occurred within otherwise intact habitats (King 1990; see O'Donnell 1996 for review). For example, in a recent study of kaka (*Nestor meridionalis*; a sympatric congener of kea), predators caused the failure of at least 30% of nests and the death of at least 20% of breeding females (Wilson *et al.* 1998, n = 20 nests over 11 years). Introduced stoats (*Mustela ermina*) are the most common carnivore in beech forests (King 1990) and they have been strongly implicated in the declines of the cavity nesting birds listed in Table 1 (Elliott 1996b; Elliott *et al.* 1996a; Wilson *et al.* 1998). Also present in kea habitat are cats (*Felis catus*), ferrets (*Mustela furo*), hedgehogs (*Erinaceus europaeus*) and brush tail possums (*Trichosurus vulpecula*) (King 1990). Kea nest on the ground (Forshaw 1989) and their nest entrances are quite large. Larger mammals consequently have access to kea nests (i.e. possums, cats and ferrets) as do those who do not usually climb trees (i.e. hedgehogs, cats and ferrets). Predator control to protect populations of mohua (*Mohoua ochrecephala*, Table 1) has increased nesting success in some places (O'Donnell *et al.* 1996) and all hole-nesters living on predator-free offshore islands enjoy high nesting success (for example, Moorhouse 1990). Although no formal monitoring of kea abundance has taken place, there is strong evidence to suggest that kea abundance today is far below that of the late-1800s – mid-1900s. Kea are still widespread (Bull *et al.* 1985), but their density is low and trends in their population size are uncertain.

A possible explanation for the apparent decline in kea abundance is that predation and/or competition from introduced mammals has reduced recruitment and increased adult mortality such that they no longer balance each other. Several species of New Zealand birds living within intact forest reserves are threatened (or have been driven extinct) by predation and competition from introduced mammals (Clout & Saunders 1995; Holdaway 1989; Innes & Hay 1991; McLennan *et al.* 1996; O'Donnell 1996). Kea might be prone to predation, especially given that their nest cavities are situated on the ground.

**Table 1. Status of cavity nesting bird populations within South Island beech forests**

| Common name | Latin name | Type of bird | Nest-hole location | Status within SI beech forest | Reference |
|---|---|---|---|---|---|
| Kaka | *Nestor meridionalis* | Parrot | Tree | Widespread at low density, evidence for rapid decline | Bull *et al.*1985; Wilson *et al.*1998 |
| Kakapo | *Strigops habroptilus* | Parrot | Ground | Presumed extinct in SI beech forest | Merton *et al.* 1984 |
| Kea | *Nestor notabilis* | Parrot | Ground | Widespread at low density, population trend unknown | Bull *et al.*1985; Personal observations; Personal communications |
| Red-crowned kakariki | *Cyanoramphus novae-zelandiae* | Parakeet | Tree | Presumed extinct in SI beech forest | Elliott *et al.*1996a |
| Yellow-crowned kakariki | *Cyanoramphus auriceps* | Parakeet | Tree | Widespread but uncommon except in localized strongholds, probably still declining | Elliott *et al.*1996a |
| Orange-fronted kakariki | *Cyanoramphus malherbi* | Parakeet | Tree | Recorded in 4 places in last 30 years, probably still declining | O'Donnell & West 1992; Read & McClelland 1984; Taylor *et al.* 1986 |
| Mohua | *Mohoua ochrecephala* | Passerine | Tree | Range greatly reduced to localized 'stronghold' in Fiordland, remnant populations elsewhere. | see O'Donnell 1996 for review. |

If predation and competition are the cause for low kea numbers today, then kea numbers are likely to continue to decline into the future unless some form of wildlife management is initiated to reverse this trend.

This study is designed to assess the likelihood that nest predators are causing an ongoing decline in kea abundance. In doing so, we aimed to increase understanding of the influence of behaviour on extinction risks of birds within New Zealand forests. Information about comparative predation risk is an important supplement to estimates of population parameters for prioritising conservation efforts (Reed 1999). Characteristics of cavity nesting birds that could influence predation risk in beech forests are- the timing of the breeding season relative to seasonal changes in predator abundance and the location, topography, and distribution of nest sites.

Large fluctuations in stoat abundance occur within beech forests in response to beech mast-seeding events (King 1983). Mast seeding is the irregular production of huge amounts of seed and it occurs in kea habitat during autumn every 4-6 years on average (Wardle 1984). This abundance of seed leads to population explosions among the animals upon which stoats prey (birds, rodents, large invertebrates, Murphy & Dowding 1995). During non-mast years, seed of any kind is rare within beech forests and stoat prey is correspondingly rare, making beech-forest marginal habitat for stoats (Murphy & Dowding 1994). In non-mast years a small peak in stoat numbers occurs in summer (December-February) when juvenile stoats become independent (King 1983). In the summer following mast seeding, however, a large peak in stoat numbers (an irruption) occurs as the increased availability of prey enables female stoats to produce larger litters (King 1983). In the autumn and winter following an irruption stoat numbers slowly return to low levels until the next beech mast (King 1983). The time at which cavity nesting birds breed in relation to this cycle is likely to influence their susceptibility to predation by stoats.

The physical nature of nest sites can also influence predation risk within a particular habitat. Among tree-nesting mohua and yellow-crowned kakariki (*Cyanoramphus*

*auriceps*), for example, predation risk decreased with the height of the nest hole above the ground (Elliott *et al.* 1996b). Overlap in use of nest sites by more than one species has also been shown to increase nest predation rates (Martin 1996), as does high density of nests within species (Hagan *et al.* 1996; Van-Horn *et al.* 1995). High nest density may encourage predators to hunt in certain places. Predation risk among kea nests could be influenced by these factors.

This chapter specifically aims to-

(i)    Estimate the nesting success rate for kea in Nelson Lakes National Park.

(ii)   Investigate the impact of periodic increases in stoat abundance on kea nest predation.

(iii)  Determine whether certain topographical characteristics of kea nest sites can influence the risk of predation.

## Methods

### Study Area

Kea occur throughout Nelson Lakes National Park and the neighbouring Upper Wairau catchment (172°52′E, 41°53′S). Kea from Travers Valley, Wairau Valley and Six Mile Valley regularly visit Rainbow Ski Area (hereafter referred to as 'the skifield') in the head basin of Six Mile Valley and these valleys comprise our study area. These glaciated valleys are U-shaped in cross section and often have grassy flats on the valley floor (c. 600-700 m a.s.l.). The steep valley sides support continuous evergreen forest up to treeline at about 1400 m. Red Beech (*Nothofagus fusca*) and Silver Beech (*N. menziesii*) dominate the canopy from the valley floors up to about 950m, above which mountain beech (*N. solandri* var. *cliffortioides*) forms the canopy, up to the treeline. Above the treeline are mountain peaks reaching 1600-2100 m and snow-tussock grasslands (*Chionochloa spp.*) with various alpine and subalpine shrubs and herbs such as *Celmisia* spp.*, Podocarpus nivalis* and *Hebe* spp.

**Locating and monitoring nests**

We found kea nests by tracking adult kea, which we caught in the skifield carpark and fitted with two-stage radio transmitters. Transmitters were harnessed to the birds with a modified backpack made of braided nylon with a cotton weak-link to ensure that transmitters eventually fell off if we failed to recapture the bird (following the methods of Karl & Clout 1987). Transmitters weighed 35 g (3.8% and 4.2% of male and female body weight, respectively) and battery life was approximately 24 months. Transmitters were purchased from Sirtrack Limited, Private Bag 1403, Havelock North, New Zealand. We tracked the birds to their nests on foot using hand held Telonics™ TR-4 receivers with Yagi aerials. Our approach was to first approximate where males were roosting at night by triangulation from the valley floor and then to visit the general area during the day to try to sight the male on provisioning visits to his mate at her nest.

Kea often re-use their nest holes, so each August and September we checked all known holes for nesting activity. We then tracked radio-tagged birds for which we had not found a nest until either a nest was found or we concluded that the bird was not breeding. Non-breeding males sometimes led us to the nests of other kea. Laying dates were determined by observing eggs in the nest or by back calculation from approximate fledging dates, assuming an incubation period of 25 days (Jackson 1963) and a nestling period of 91 days (this study).

**Calculating nesting success**

We found nests at all stages of development from egg laying to fledging (Fig. 1) and two nests were still active when fieldwork ended in February 1999. Consequently, some nests were under observation from egg laying to fledging while others were monitored for only a few weeks. The likelihood of observing nest failure increases with the length of time over which a nest is observed (Mayfield 1961), so the 'exposure' of each nest needed to be incorporated into calculations of nesting success to avoid biased estimates. The exposure of a nest is the time (in days) from when it was confirmed to contain eggs or chicks until it either failed or the chicks fledged. Nest bowls that were obscured from view were not included in the sample until we heard chicks begging or found eggshell

remains outside. We checked each nest every two to three weeks until it either failed or the chicks had fledged.   Nests that contained dead chicks or were empty well before their expected fledging date were regarded as failures. The outcome of nests that were found empty within two weeks of their expected fledging date was determined by observing the behaviour of the adults and searching for fledglings.  The presence or absence of fledglings is easily confirmed by observing the adults and visiting the treeline directly above the nest, where the fledglings remain for two to six weeks to be fed by their parents. The day of failure or fledging was estimated as the midpoint between the date when the nest was found empty and the date of the previous visit (Mayfield 1961).

Daily nest survival for a sample of nests is calculated as the total number of failures divided by the total number of days exposed (Mayfield 1961). Overall nest survival is then estimated from the daily nest survival rate by raising it to the power of the length of the nesting period (egg laying to fledging) (Johnson 1979), which is 116 days for kea . The variance of a daily survival estimate was calculated using Johnson's (1979) formula $((\text{exposure} - \text{losses}) \times \text{losses})/ (\text{exposure})^3$. The standard error of the estimate is the square root of the variance and 95% confidence limits are the estimated value $\pm 2$ standard errors (Johnson 1979).

**Figure 1. Stage in the breeding cycle at which kea nests were found.**



**Age of nest when found (days)**
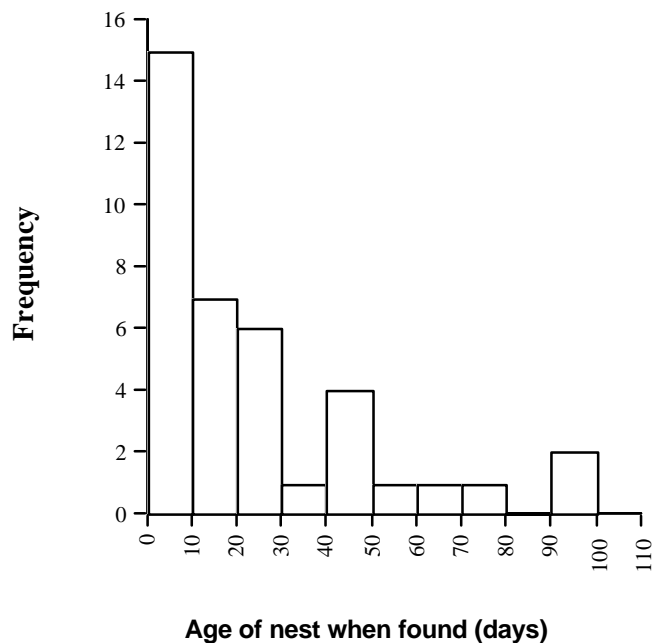
Among-year comparisons in daily nest survival rates were made using one-way analysis of variance (ANOVA). To do this, we generated a set of daily survival rates for each year by random sampling from a normal distribution with mean and standard deviation as estimated for that year.

**Beech seeding and stoat abundance**

Annual beech seed production data for the Nelson Lakes area was obtained from Wilson *et al*. (1998) for 1992-1996 and from Department of Conservation (unpublished data) for 1997-1998. Both of these sources collected seed using the methods described in Wilson *et al*. (1998). Beech seedfall is expressed in terms of seeds/m$^2$ and we use Wardle's (1984) definitions of a full mast (>4000 seeds/m$^2$), partial mast (500-4000 seeds/m$^2$) and poor mast (<500 seeds/m$^2$). An index of relative stoat abundance for 1992-1995 was obtained from Wilson *et al.* (1998). No stoat abundance data was available for 1995-1998, but the relationship between beech seedfall and stoat abundance is well enough known (King 1983; Wilson *et al.* 1998) for general inferences about stoat abundance in 1997 and 1998 to be made from the beech seedfall index.

**Nest site topography and nest predation**

We described the topography of each nest site in terms of altitude a.s.l. and slope. We measured the gradient of the slope from 50 vertical metres above the nest to fifty vertical metres below it, using NZMS 260 1:50 000 topographical maps. Expressing slope in this way is a simplification of the real situation, because slope also influences the amount of exposed rock, size of trees etc. Steep nest sites tended to be rockier and generally less hospitable than nest sites on gentle slopes. Slope as we have expressed it, however, matches our subjective impression of the relative topographies of nest sites and simplifies the statistical analysis. We measured altitude with a portable altimeter or estimated it using prominent landforms and NZMS 260
 1:50 000 topographical maps.

To determine the effect of altitude and slope on nest predation rates, we fitted a series of logistic regression models to the data. Model selection was based on Akaike's

Information Criterion (AIC; Lebreton *et al.* 1992), with the best models having a low AIC value and a high model weight calculated from the differences in AIC (Buckland *et al.* 1997). We examine the significance of specific terms included in the models using likelihood ratio tests (LRT). This approach involves calculating the difference in –2.log likelihood values between a model that includes the term in question and one that does not. The resulting statistic follows a chi-square distribution with one degree of freedom (Hosmer & Lemeshow 1989).

## Results

We monitored 36 kea nests between 1993 and 1998 (Fig. 2). In 1998 we were able to find more nests because more birds were radio-tagged and because some unmated radio-tagged males led us to previously unknown pairs on the periphery of the study site. We failed to find any nests in 1996, despite visiting all known nest holes three times over the spring, and despite an intensive radio tracking effort. Nests were found at all stages of the breeding cycle from egg laying until just before fledging, although 74% were found before the eggs hatched (Fig. 1).

**Figure 2. Kea nests found at Nelson Lakes from 1993-1998.**
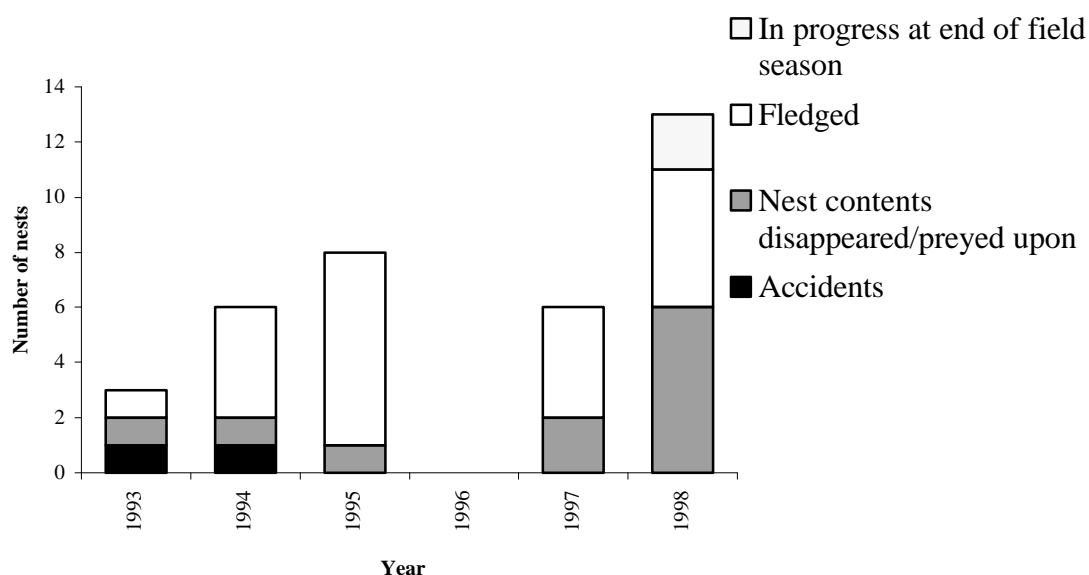
## Nest density

The density of kea nests within our study site was one per 4.4km$^2$. No two nests were situated on the same spur (i.e. above and below each other); adjacent nests were separated by 750 – 1750 m.

## Timing of the breeding season

Laying dates were estimated for 26 nests (Fig. 3). Breeding began in August and was completed by mid-January in all years except 1998, when the season was extended by 3 re-nests. Two of the re-nests followed early failure of the first nests and the other re-nest followed the successful fledging the pair's first nest. These are the first records of double brooding among kea.

**Figure 3. Estimated laying and fledging dates of 26 kea nests. π= date of lay, o = date of fledge.**
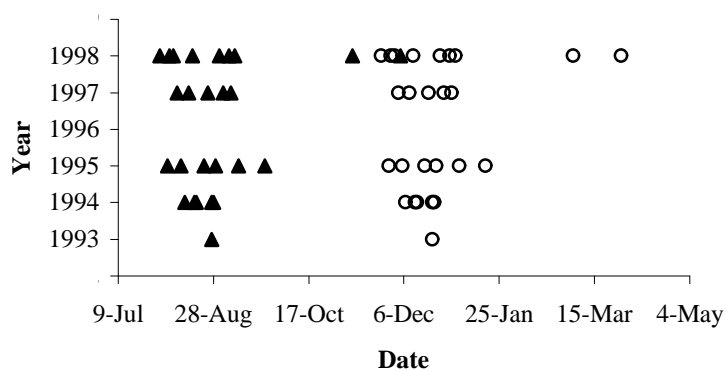


## Nest fates

Of the 36 nests that we monitored, 57% fledged at least one chick, 6% failed by accident (the chicks fell down a large crevice in their nest hole) and 6% were still in progress when observations finished (Fig 2). The eggs or chicks disappeared from the remaining

31% of nests. At one of these nests one of the nestlings had previously been radio-tagged and we recovered its carcass from a stoat den about 100m below the nest. Tooth marks on the transmitter were consistent with stoat dentition and the dead chick's nest-mate was missing. In further data analyses, we include this nest as one from which the chicks mysteriously disappeared, as this would have been the case had the chick not been radio-tagged. At no period in the nesting cycle were nest contents particularly likely to disappear (Fig. 4).

**Figure 4. Stage in the breeding cycle at which nest contents disappeared.**
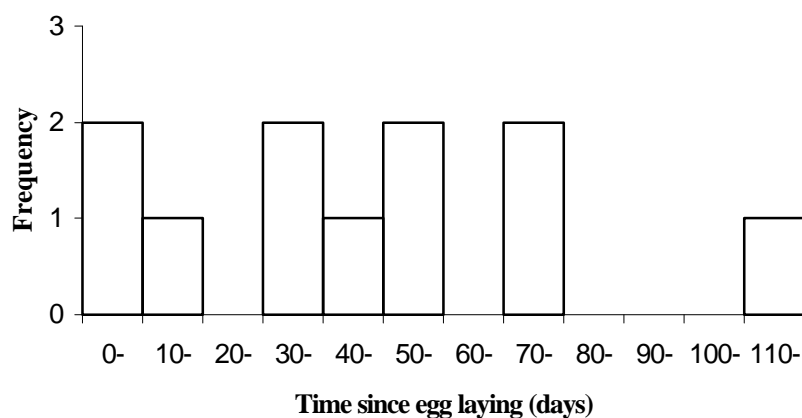


**Overall nest survival**

The overall daily nest survival rate excluding accidental failures was 0.995 (SE = 0.001), equivalent to at least 57% (95% CI = 40-79%) of nests avoiding nest predation. If accidental failure occurs at 6% of nests (see above), then the overall nest survival rate will be 51% (95%CI = 34-73%).

**Beech seedfall and stoat abundance**

A full beech-mast occurred in autumn 1995, poor seeding occurred in 1993, 1996 and 1998 and there was negligible seeding in the other years (Fig. 5). The only stoats Wilson *et al.* (1998) captured during our study were caught in August and November following the 1995 seed fall (Fig 5). We assume that stoat numbers peaked in late summer 1996 and were at very low levels in 1997 and 1998 (King 1983; O'Donnell et al. 1996; Wilson et al. 1998).

**Figure 5. Response of stoat numbers (---) to beech seedfall (histograms) in Nelson Lakes National Park.　 = stoat catch data from Wilson *et al.* (1998). Stoat abundance after November 1995 are hypothetical, based on previous studies of stoat response to beech seedfall by King (1983), Wilson *et al.* (1998) and O'Donnell *et al.* (1996).**
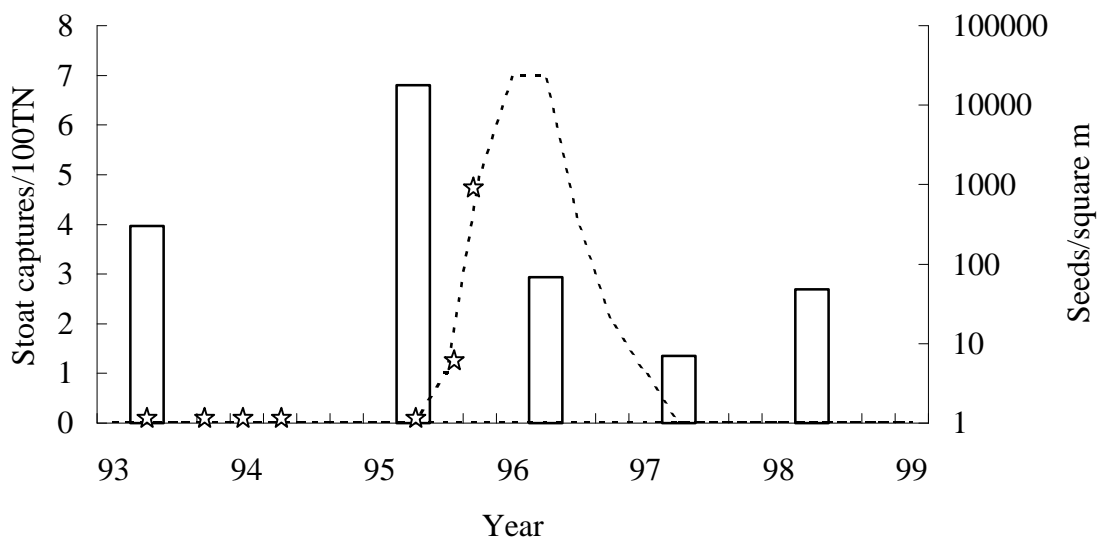


**Among-year differences in nest survival**

There were significant among-year differences in daily nest survival (ANOVA, F = 140.46, P<0.000) (Fig.6). Nest survival did not decrease in 1995 when stoat numbers were elevated after the beech mast.

**Figure 6.  Among-year differences in daily nest survival.  Error bars are standard errors.  The  horizontal lines indicate pairs of years with significantly different daily survival rates as determined using Tukey's pairwise comparisons.**
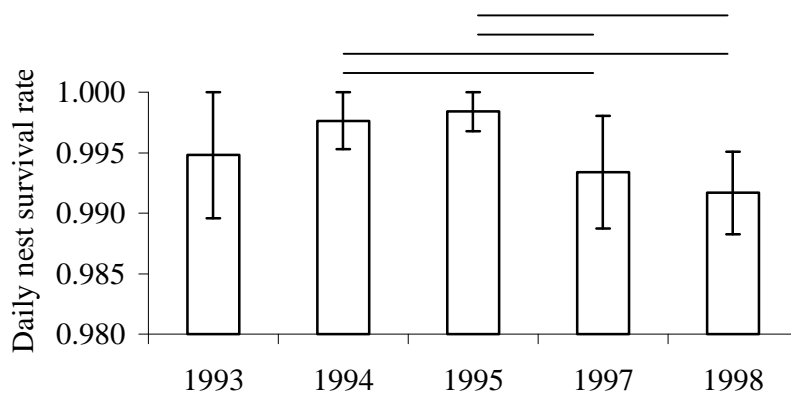


| No. of nests predated | 1 | 1 | 1 | 2 | 6 |
| --- | --- | --- | --- | --- | --- |
| Exposure days | 192 | 424 | 620 | 303 | 720 |

## Use of nest sites

The 36 nests occurred at 22 nest sites (Fig. 7).  Of the 36 nests, 50% occurred at sites in which we had previously observed nests.

**Figure 7.  Use of nest sites.**
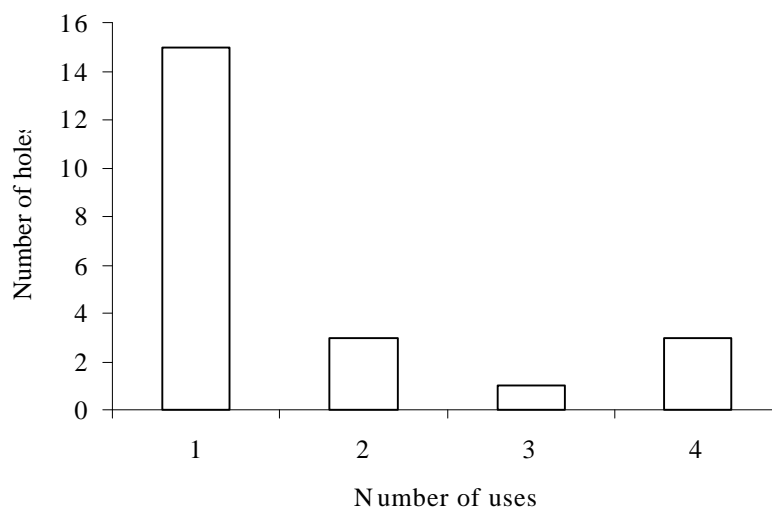
**Nest site topography**

Nest sites were situated at altitudes ranging from 720–1320 m a.s.l., with 90% of nests situated at over 900 m (Fig. 8). Slope at nest sites ranged from 25-60 degrees and there was no obvious tendency toward either extreme (Fig. 9).

**Figure 8. Altitude at nest sites.**



Altitude (metres a.s.l.)
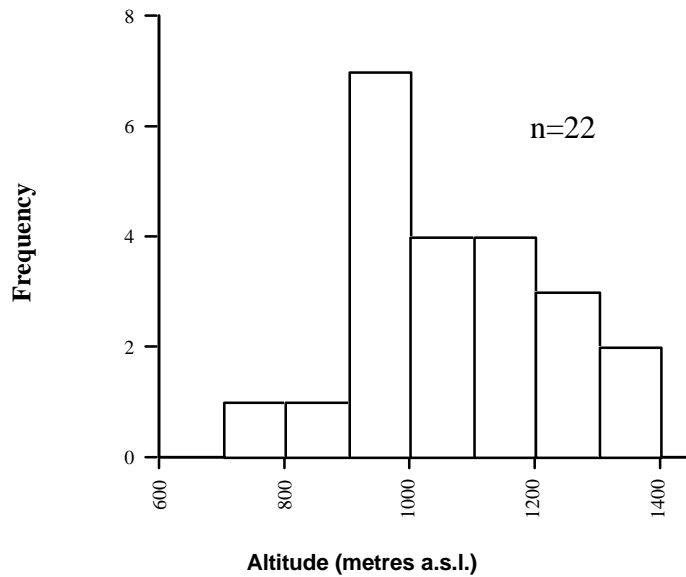
**Figure 9. Slope at nest sites**



S lope at nest site (degrees)
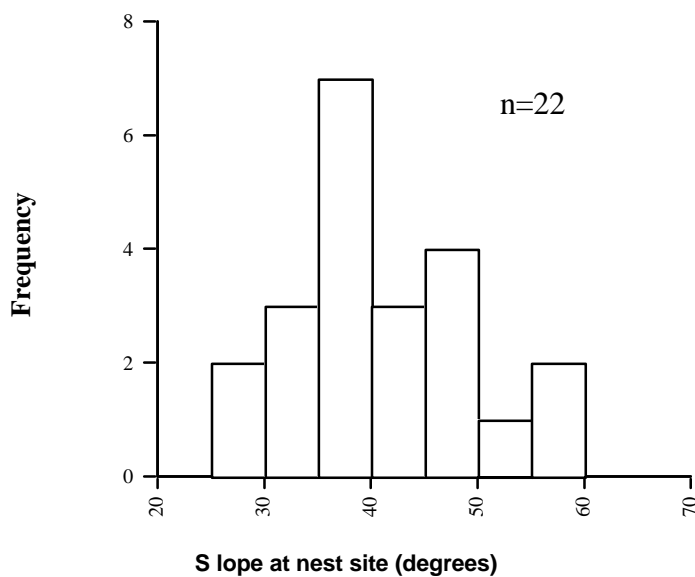
To investigate relationships between nest site topography and the probability of nest contents disappearing, we pooled the data from the different years into a single data set (Figure 10). Nest sites that were used in more than one year were included only once to avoid pseudo-replication in case something we did not consider was determining their predation risk. These nest sites were counted as 'failed' if the nest contents disappeared in at least one of the years in which they were used.

**Figure 10.   Nesting success in relation to nest site topography, for all years pooled. ▲= failed sites, ○ = successful sites.**
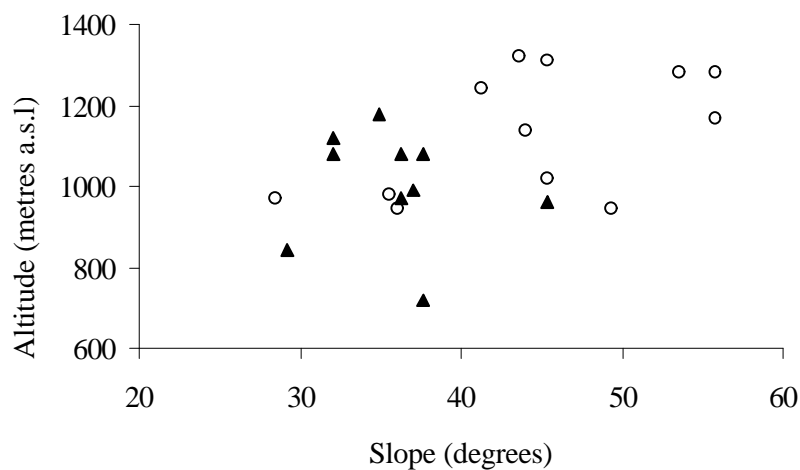


Model selection using AIC failed to identify a single best model (Table 3). Therefore, we used Likelihood Ratio Tests to examine the importance of specific model terms. There was a significant slope effect ($\chi^2 = 4.77$, df = 1, P = 0.029) but no significant altitude effect ($\chi^2 = 0.954$, df = 1, P = 0.329) (Table 4). There was no support for an interaction between altitude and slope ($\chi^2 = 3.038$, df = 1, P = 0.081). Altitude may also have proved significant had we been able to monitor more nests.

**Table 2. Model-fitting summary for logistic regression models fitted to nesting success data. ΔAIC is the difference in the AICs of the current model and the model with the smallest AIC.**

| Model | -2.log likelihood | df | AIC | ΔAIC | Model Weight |
|---|---|---|---|---|---|
| Slope | 22.118 | 2 | 26.118 | 0.000 | 0.365 |
| Altitude | 25.934 | 2 | 29.934 | 3.816 | 0.054 |
| slope + altitude | 21.164 | 3 | 27.164 | 1.046 | 0.217 |
| slope + altitude + slope*altitude | 18.126 | 4 | 26.126 | 0.008 | 0.364 |

**Table 3.  Likelihood ratio testing (LRT) for the significance of specific model terms. Deviance scores for each effect are the difference in –2*log-likelihood values between a model that includes the term in question and one that does not.**

| Effect | df | Δ Deviance | P |
|---|---|---|---|
| slope | 1 | 4.77 | 0.029 |
| alt | 1 | 0.954 | 0.329 |
| Alt*slo | 1 | 3.038 | 0.081 |

This result is not the result of a greater exposure at the failed sites.  In fact, exposure at the steep/high successful sites was greater than at the failed sites, and 4 of the 7 sites that were used multiple times never failed, including 2 of the 3 sites that were used four times and the site that was used three times.

Neither slope nor altitude were significant predictors of nest survival in logistic regression models for individual years (Ps >0.05), but the effect of topography appears to have been particularly strong in 1998 (Fig. 11).

**Figure 11.  Nesting success in relation to nest site topography for individual years.**

**X-axes = slope (degrees from horizontal), Y-axes = altitude (metres above sea level).**

**▲ = failed sites, ○ = successful sites.**
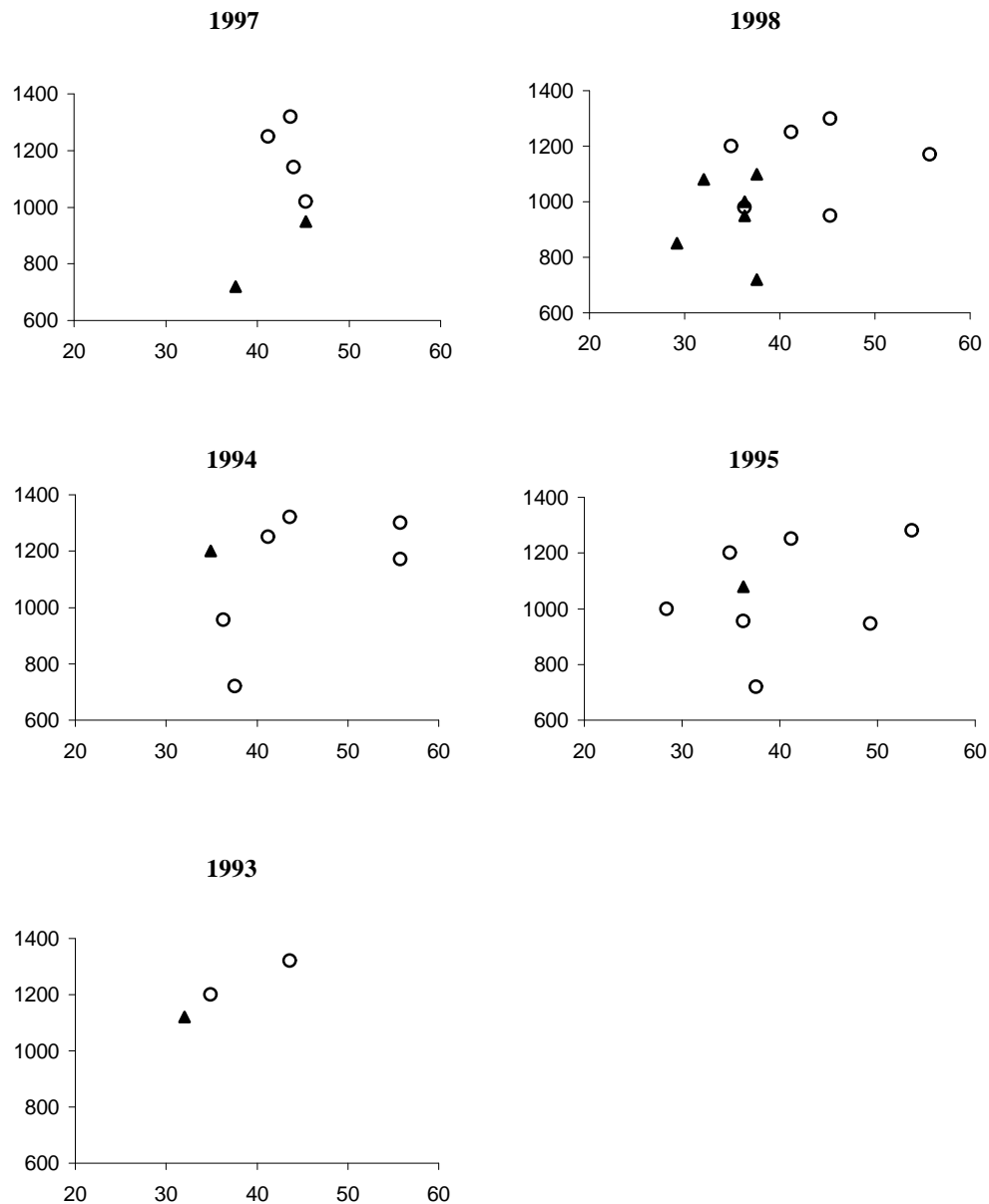
**Predation of adult females in their nests**

The females from 5 of the 11 nests whose contents disappeared had been marked with leg bands prior to the disappearance of the eggs or chicks.  All of these were confirmed to be alive after the nest contents disappeared.  At the other six nests where the nest contents disappeared, no adult females were found dead in the nests.

## Discussion

**Overall nesting success**

Our estimate of 51% of kea nests fledging at least one chick is similar to 54% for yellow-crowned kakariki (Elliott *et al.* 1996a) and 55% for mohua (Elliott 1996a) in population 'strongholds' in Fiordland.  In contrast, kaka nesting success (in tree cavities) near our study site between 1985 and 1996 was only 10% and at least 30% of these kaka nest failures were attributed to stoats (n= 20 nests, Wilson *et al.* 1998). Being on the ground, then, does not seem to predispose kea nests to predation.

Steep, higher altitude sites might be over-represented in our sample because they were more likely to survive to the dates on which we searched for nests (Green 1977).  Such a bias would result in an over-estimate of nesting success. Overestimation can only be slight, however, as 75% of nests were found during incubation (Fig. 2). Moreover, Johnson (1979) shows that the Mayfield estimator is fairly robust to this kind of heterogeneity among nests.

**Predation of kea nests**

Even if all of the nest failures we observed were caused by predators, the predation is too low to justify immediate predator control to enhance nesting success among kea.  This is especially so given that adult females seemed to escaped most nest failures.  Jackson (1963) observed 17 kea nests with chicks in Arthur's Pass and did not write of finding chicks or nesting females missing or preyed upon. Unfortunately he did not present an estimate of nesting success that can be compared with ours, but he gave the overall impression that nest predation was uncommon during his study.  Nowhere in his 1963

paper does he specifically address nest predation as an issue for kea, and of rats and stoats he states 'I have found no evidence of these animals affecting Keas' (Jackson 1969, p44).

Although predation was confirmed as the cause of failure at only one nest, we believe that predators also caused many of the unexplained nest failures. The significant correlation between slope and nest failure supports this conjecture, as we can hypothesise how slope might influence predator movements, but not how slope would influence any of the other potential agents of nest failure, such as food shortages, disease, nest parasites and parental inexperience.

**Among-year differences in nest survival**

As only a small number of nests were found each year and nest site topography influenced survival probability, daily survival estimates for individual years are likely to be strongly influenced by the particular sites in which the birds chose to nest in that year. We are, therefore, cautious about making biological interpretations of the differences in daily nest survival, especially since the species of predator was unidentified in most cases. However, the nests that we monitored in 1995 (after beech mast-seeding) were spread across a range of topographies (Fig. 11) and nesting success still did not decrease significantly. This result contrasts with the general pattern of mast seeding being detrimental to avian nesting success (King 1983; O'Donnell *et al.* 1996; Wilson *et al.* 1998). Jackson (1963) also did not report an increase in nest predation in the breeding season following heavy beech seeding in the Arthur's Pass region in 1960 (New Zealand Forest Service unpublished data). We strongly recommend, however, further monitoring of kea nests in years following beech masts because of the small numbers of nests that were monitored in these studies.

**Timing of the breeding season**

Breeding among kea was generally over by mid-January each year. Very few nests were active in late-January and February, which is when juvenile stoats usually become

independent and stoat numbers become elevated (King 1983). Mohua and kakariki both nest in late summer and their nest predation rates significantly increased during stoat population irruptions (Elliott 1996a; Elliott *et al.* 1996a). Kaka also nest in late summer (December – February; Wilson *et al.* 1998). This is probably part of the reason why predation of kea nests was relatively low and did not increase significantly in 1995. Incidental observations of kea that lacked access to a human food source showed that early nesting also occurs among these birds, sometimes as early as nesting among the earliest 'skifield' pairs. Over the years we found three 'non-skifield' nests and saw new fledglings from six others. In all of these cases, the eggs were laid in late-July or August. Breeding was less synchronized during Jackson's (1963) study than in ours, but the peak was still well before January (Fig. 12). The reasons for kea breeding so early relative to other birds is not clear. It is difficult to see that the skifield could have advanced the average date of laying to the extent that nest survival in our study site was substantially higher than it is in the rest of the kea population. This is especially so given the protection afforded to many kea nest sites by their topographical characteristics.

**Figure 12. Timing of kea nesting activity at Nelson Lakes (1993-1998) and at Arthur's Pass (1956-1963: Jackson 1963).**
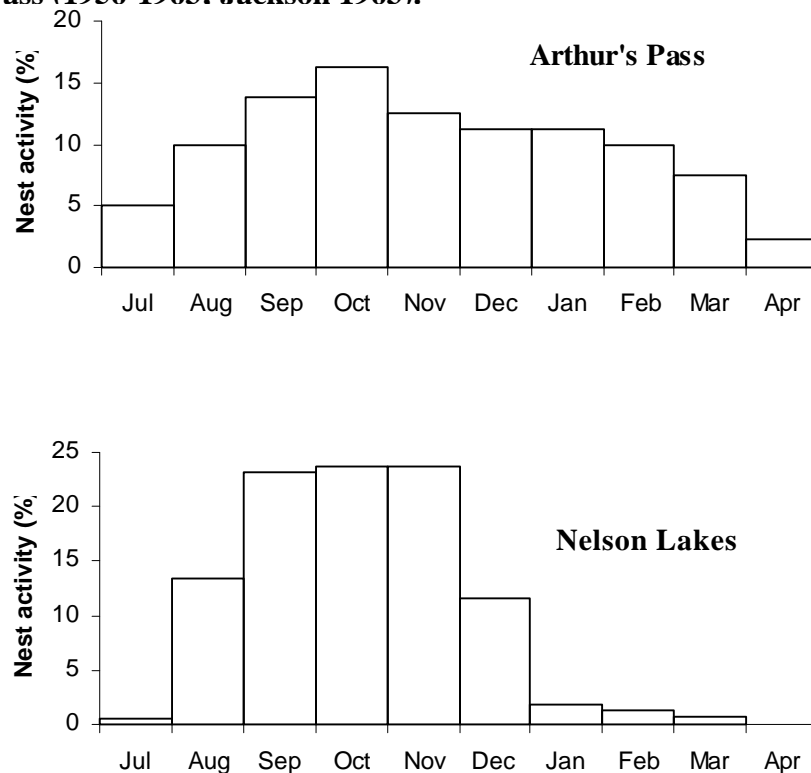
**Nest density and nest site topography**

Predators of kea nests in our study area seem to avoid the steepest places. As most of the predator species present are quite capable of travelling in steep places, it appears that they do not identify them as a source of prey and so hunt elsewhere. If the density of kea nests was high, then predators might learn to hunt steep areas and the kea nest predation rate would accordingly be higher. In previous North American studies, nest predation rates increased with increased density of nests (Keyser *et al.* 1998; Martin 1996). In contrast to the situation with kea, kaka nests can occur within 100m of each other (Peter Dilks unpublished data) and the location of their nest sites is similar to that of kakariki (*Cyanoramphus* spp.), grey warbler (*Gerygone igata*), riflemen (*Acanthisitta chloris*) and, in some places, mohua. Suitable nest cavities for these species are concentrated at low altitude, where trees are larger, leading to a high density of tree nests here. This, combined with the strong association between slope and nest survival, could help to explain the difference in nest predation rates between kea nests and kaka nests which occurs despite kea nests being situated on the ground. The persistence of takahe (*Notornis mantelli,* a ground-nesting, flightless rail) in the Murchison Mountains and Hutton's Shearwater (*Puffinus huttoni,* a burrow-nesting petrel) in the Kaikoura Ranges, may be due to similar benefits of being one of few bird species nesting in a rugged landscape at high altitude in New Zealand.

**Predation of adult females in their nests**

The apparent high survival rate among adult female kea whose nests failed may be due to the nature of the predators that took the nest contents (if it was predators) or to the nature of kea nest cavities, or both. Adult females are unlikely to be attacked by a rat, possum or hedgehog, so these animals could have been responsible for the nest failures we observed. Kea nest cavities have between one and eight square metres of floor space, and many have multiple entrances, whereas the nest holes of kaka, kakariki and mohua (in trees) typically have one entrance above nest level. If the female kea gets off her brood when she hears or sees a predator in the entrance tunnel, she has a chance to escape, particularly if the brood distracts the attention of the predator from the female. In a tree nest, an entering predator would probably drop straight onto a brooding female. This

could help to explain the difference in adult female survival between kea and kaka, if this difference is indeed real. A high escape rate among adult females greatly reduces the impact of nest predation on population growth (see chapter 3).

## Conclusions

Due to the relatively low kea nest predation rate and the apparently regular escape of adult females, we conclude that there is no immediate need for predator control to enhance kea nesting success. However, we strongly recommend that a larger sample of adult females be marked (preferably radio-tagged) in order to increase confidence in this conclusion. The natural 'safeguards' to nest predation that we have identified may operate in most kea habitat. It would be desirable to replicate this study in another South Island site to see if this is so. Logistical considerations would make another nesting study a long-term commitment. The 'safeguards' may also have the effect of making predation rates unlikely to change in the future, as long as the suite of predators remains the same as it is now. Our results suggest that the current low density of kea either reflects a return to a pre-European state (i.e. that kea are naturally rare) or that kea numbers are slowly declining through more subtle mechanisms such as competition for food combined with a low level of predation. Chapters 3 and 4 address this issue further. Our results help to explain kea's persistence despite the legal killing of approximately 150 000 kea between the late 1800s and 1986, when they were classified as a pest to agriculture (Anderson 1986). If kea nests were also suffering high rates of predation during this time, then kea would probably presently be either extinct or endangered with extinction.

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# 3. Kea (*Nestor notabilis*) population dynamics in Nelson Lakes National Park

## Abstract

Inference about trends in the size of the wild kea (*Nestor notabilis*) population is desirable owing to their small population size, their taxonomic distinctiveness and the potential threat placed on them by introduced mammals. Monitoring of kea populations using abundance indices has been prevented by unpredictability in the conspicuousness of individuals and their sparse distribution within a rugged and inaccessible habitat. This study takes an alternative approach to making inferences about population trends. This approach involves estimating vital rates and investigating the impact of conservation threats such as introduced predators on these parameters. Age specific survival and adult productivity were studied among a sample of wild kea using radio-telemetry. Adult female survival was estimated at 82% per annum and productivity was estimated at 0.33 female chicks per adult female per year. One year of fledgling survival data suggests that a high proportion of fledglings survive their first year of life in at least some years. A matrix population model is presented and used to conduct a sensitivity analysis aimed at measuring the relative importance of each demographic parameter to population growth. A change in adult survival has roughly four times the impact on population growth as a proportionally equivalent change in any other parameter. As adult survival was high, the kea population is unlikely to be in rapid decline. Introduced mammalian predators had very little impact on any parameter, so it is unlikely that they are causing a rapid decline in kea population size. Kea numbers could, however, be declining through a combination of competition for food from browsing mammals and a low level of predation, but more research is required to test this hypothesis. An outline of proposed future research is presented that will increase confidence in conclusions about the conservation status of the wild kea population.

## Introduction

Kea (*Nestor notabilis*) are large (males = 930g, females = 830g) green parrots endemic to the montane areas of New Zealand's South Island. They are one of two extant members of the genus *Nestor* which is the sole genus in the New Zealand sub-family *Nestorinae* (Forshaw 1989). Taxonomically distinctive traits of kea include nesting on the ground, frequenting the alpine zone, exhibiting a sexual size dimorphism and exhibiting a high degree of sociality (Diamond & Bond 1991; Forshaw 1989; Jackson 1963).

Obtaining a reliable estimate of current population size is very difficult for logistical reasons. A commonly cited estimate of 1000-5000 individuals (Anderson 1986) is little more than a guess based on extrapolating density estimates from a few localized areas to the rest of the South Island. Whatever the exact size of the population, it is certainly small on a world scale and probably small in relation to its own history. An estimated 150 000 kea were legally destroyed between 1880 and 1986 (Anderson 1986) when kea were considered to be a threat to sheep farming because of their habit of digging into the backs of live sheep to feed on perirenal flesh (Benham 1907). They received full protection in 1986 (Anderson 1986).

With European colonization of New Zealand over the last two centuries came the invasion of a host of introduced mammals into kea habitat (King 1990). Predation and competition from these mammals has been strongly implicated in the declines and extinctions of many previously abundant New Zealand bird species (Clout & Saunders 1995; Holdaway 1989; Innes & Hay 1991; McLennan *et al.* 1996; O'Donnell 1996). Conservation management has been urgently required to save some species from extinction. The impact of introduced mammals on kea population dynamics has never been formally investigated, although two previous studies of aspects of kea population biology suggested that kea numbers in Arthur's Pass are relatively stable (Bond & Diamond 1992; Jackson 1960; Jackson 1963; Jackson 1969). However, failure to distinguish between dispersal and mortality, combined with the unknown effects on kea movements of human food sources such as the refuse dump within the study area, left a large amount of uncertainty about population trends. Recent advances in radio-tracking

technology now enable detailed studies of individual population parameters, such as survivorship and productivity to take place, thereby giving the opportunity to clarify the conservation status of kea.

Radio telemetry is expensive and so requires careful study design to maximize conservation gains for each dollar spent. Some population parameters have more impact on population growth than others and these should be the main focus of conservation studies. Conducting sensitivity analyses on models of population growth can identify the most important population parameters. Sensitivity analysis is an objective assessment of the relative effect of changes in demographic parameters on the population growth rate or probability of extinction (Caswell 1978). Seal *et al.* (1993) conducted a sensitivity analysis for kea, using rough population parameter estimates from Jackson's (1960, 1963 &1969) papers combined with observations of captive kea and educated guesses. They showed that research should probably focus on investigating age-specific mortality rates (especially in females), productivity, and the proportion of adults breeding in any one year (Seal *et al.* 1993). Our research programme, which uses extensive radio telemetry, began at Nelson Lakes National Park in 1992 as a result of these recommendations.

Even with radio-telemetry as a tool, we anticipated great logistic difficulty in a kea population study, so we began with a study area in which most juveniles and adult males scavenge food from a skifield carpark for three months of the year (August – October). This had significant logistic and technical advantages, which enabled us to quickly develop radio-tracking techniques and rapidly answer questions about nest predation (presented in Chapter 1). In this paper, we present data on adult survival, productivity and preliminary data on juvenile survival. This data is integrated into a contemporary analysis of kea population dynamics which we use, along with technical knowledge and budget considerations, to recommend a course of action that will maximize the future efficiency of kea population research.

## Methods

### Study Area

Kea occur throughout Nelson Lakes National Park and the neighbouring Upper Wairau catchment (172°52′E, 41°53′S). Kea from Travers Valley, Wairau Valley and Six Mile Valley regularly visit Rainbow Ski Area (hereafter referred to as 'the skifield') in the head basin of Six Mile Creek and these valleys comprise our study area. These glaciated valleys are U-shaped in cross section, often with grassy flats on the valley floor (c. 600-700 m a.s.l.).  The steep valley sides support continuous evergreen forest up to treeline at about 1400 m. Red Beech (*Nothofagus fusca*) and Silver Beech  (*N. menziesii*) dominate the canopy from the valley floors up to about 950m, above which mountain beech (*N. solandri* var. *cliffortioides*) forms the canopy, up to treeline.  Above treeline are mountain peaks reaching 1600-2100 m and snow-tussock grasslands (*Chionochloa spp.*) in which grow various alpine and subalpine shrubs and herbs such as *Celmisia* spp*., Podocarpus nivalis* and *Hebe* spp..  Lake Rotoiti and the skifield road allowed access to the valleys within the study area.

### Radio-telemetry

Radio-telemetry was the primary tool with which we found nests and followed the fates of individuals.  We used two-stage radio-transmitters each weighing 35 g (3.8% and 4.2% of male and female body-weight, respectively) and with battery life of 24 months. Transmitters were purchased from Sirtrack Limited, Private Bag 1403, Havelock North, New Zealand.  We captured kea in the skifield carpark and fitted them with transmitters using backpack type harnesses made of braided nylon.   A cotton weak-link was incorporated into each harness so that the transmitter would eventually fall off if we failed to recapture the bird (Karl & Clout 1987).  Using hand-held Telonics [TM] TR-4 receivers with yagi aerials, we were almost certain to detect a signal if we were in the same valley as the bird and within one-two km upstream or downstream.

**Parameter estimates**

*Productivity*

We define productivity as the number of female chicks successfully fledged per adult female per year. We consider in detail several parameters that combine to determine productivity. These are the frequency of breeding years, the proportion of adult females that attempt to breed in breeding years, nest survival, the number of fledglings emerging from successful nests and the fledgling sex-ratio. We could not gather enough data on clutch size and hatching success to estimate these parameters. Nest survival estimation methods and results are presented in Chapter 1 but the other estimates are original in this chapter.

*Fledgling sex ratio*

We determined the gender of fledglings using both molecular and morphometric methods. We collected 0.4mls of blood , via syringe, from the brachial wing vein of all chicks fledged from five nests in 1997 (n=11 chicks). Blood was stored under refrigeration at 1-4 degrees centigrade in cryotubules of lysis buffer before being flown to McMaster University in Canada for DNA sexing by Dr Jim Quinn using the methods of Kahn *et al.* (1998). A sexual size dimorphism can be used to reliably determine gender among adult kea (Bond *et al.* 1991). We investigated whether the same can be done for fledglings by plotting head size against upper-mandible length for thirteen fledglings whose gender had been determined in the molecular analysis. If morphometric sexing was possible, then we would use it to sex additional chicks from 1998 nests.

*Adult survival*

We classify kea as 'adult' when they are three years old. At this age, their juvenile markings disappear (this study, n=5) and they are sexually mature by three and a half years (this study, n=2).

We radio-tagged 37 different adult kea (8 females and 29 males) between 1992 and 1998. We measured survival in annual terms, between successive Octobers as this is when we usually caught birds and this period entirely encompasses peaks in stoat (*Mustela ermina*)

abundance caused by heavy beech seeding (Chapter 1; King 1983). We only recorded a death if a carcass was found; the disappearance of a bird was not regarded as a death. The intensity of our radio tracking was such that we are confident that the disappearance of birds was due either to transmitter malfunction, flat batteries or emigration, rather than to dead birds falling down holes and being undetectable. Most radio-tagged individuals were monitored for more than one year, giving a total sample of 89 kea-years (70 male and 19 female). We calculated survivorship by dividing the number of deaths by 89 and subtracting this from 1.

*Adult female survival*

All kea in our study area were monogamous. If the sex ratio is uneven in a monogamous population then it is important to estimate the survivorship of the rarer gender rather than the more common one. Adult female kea perform all incubation and brooding, which makes them potentially more prone to mortality by predation than males (Elliott 1996b; Wilson *et al.* 1998). It is, therefore, likely that adult female survival is lower than adult male survival and that the sex ratio is accordingly skewed toward males. Unfortunately, logistical considerations prevented us from radio-tagging enough adult females to separately estimate their survival from telemetry data. Instead, we assume that somatic (non-reproductive) survival among adult females is equal to that measured from the telemetry data (above), and that overall adult female survival equals somatic survival minus nesting mortality. We aimed to measure nesting mortality by colour banding adult females so that we could tell whether they went missing when their nests were preyed upon.

*Juvenile survival*

We radio-tagged eleven chicks at fledging time in the 97/98 summer. In January 1999 (one year later) we flew at 7-10 thousand feet, in a radio-telemetry equipped fixed-wing aircraft (Cessna 172), over all forested mountains within a 50 kilometre radius of the nests. Following the flight, we visited all of the birds on foot to see if they were alive. We are confident that we were just as likely to detect a dead bird as a live one, as reception and coverage from the plane was exceptionally good. For example, one

transmitter was buried under two-metres of rock scree and we detected its signal from over two kilometres away. We therefore censored missing birds from the dataset.

**Population Dynamics**

We define a matrix population model, based on the basic form $\lambda$ = Adult survival + (Productivity x pre-adult survival), where $\lambda$ is the population growth rate. The model we use for the kea population is $\lambda$ = Adult survival + (Frequency of breeding years x Proportion of adult females laying in breeding years x Nest survival x Fledglings per successful nest x Fledgling survival x Juvenile survival x Sub-adult survival), where fledglings are kea aged 0-1, juveniles are 1-2, and subadults are 2-3.

The sensitivity of $\lambda$ with respect to any demographic parameter, $x_i$, is defined as $d\lambda/dx_i$ (Caswell 1989). Sensitivity indices are not directly comparable between parameters, as units differ between parameters. To enable direct comparison, we calculated elasticity indices ($E_i$) for each parameter. $E_i$ is a measure of the effect of a proportional change in the parameter on $\lambda$ and is calculated as $E_i = dLn\lambda/dLn(x_i) = (1/\lambda) (d\lambda/dx_i)x_i$ (Kroon *et al.* 1986). Elasticity indices are meaningful relative to each other. For example, a 10% change in a parameter with $E_i = 0.8$ has 4 times as much impact on population growth as a 10% change in a parameter with $E_i= 0.2$.

## Results

**Productivity**

*Frequency of breeding years*

Most of the radio-tagged males bred in five of the six years between 1993 and 1998. We found no nests in 1996, despite a good ski season giving adult males their usual access to high-energy food in the skifield carpark. The absence of breeding in 1996 was confirmed in winter 1997, when fledglings were absent from the skifield carpark (Figure 1). Fledglings were also absent from the skifield carpark in 1992 when the study began, suggesting that 1991 was also a non-breeding year. This could suggest a minimum breeding frequency of five years out of seven and a maximum of four years out of five.

Our best guess is the midpoint of this interval, which is 76% (Table 1), or approximately three out of four years.

**Figure 1. Counts of fledgling kea at Rainbow Skifield.  No fledglings were seen in 1992 or in 1997.**
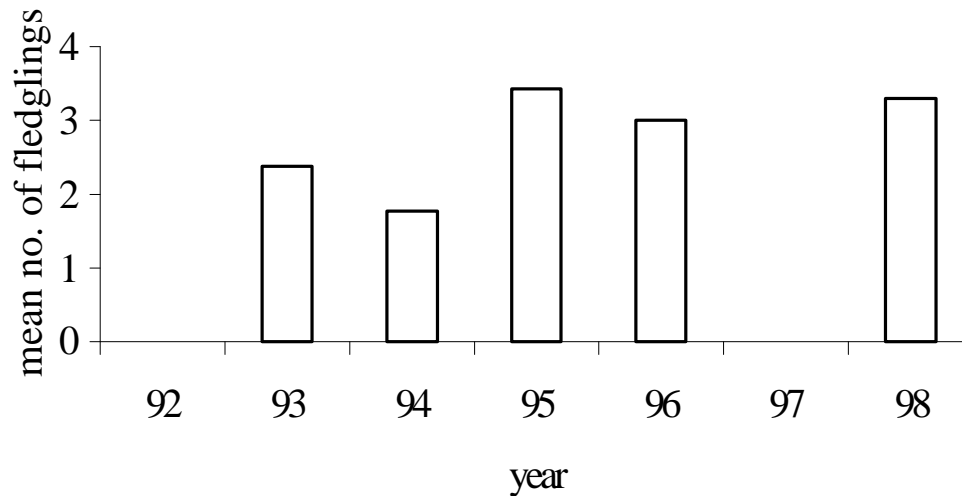


**Table 1.  Estimates of each parameter with sensitivity ($S_i$) and elasticity ($E_i$) indices.**

| Demographic parameter | Estimate | $S_i$ | $E_i$ |
| --- | --- | --- | --- |
| Adult survival | 0.82 | 1.00 | 0.87 |
| Fledgling survival | 0.60 | 0.21 | 0.13 |
| Nest survival | 0.51 | 0.25 | 0.13 |
| Frequency of breeding years | 0.76 | 0.17 | 0.13 |
| Juvenile survival | 0.75 | 0.17 | 0.13 |
| Subadult survival | 0.85 | 0.15 | 0.13 |
| Fledging success | 0.93 | 0.14 | 0.13 |
| Proportion breeding | 0.93 | 0.14 | 0.13 |

*Proportion of adult females breeding in breeding years*

In breeding years, 93% of territory holding pairs laid eggs (n=43 pairs; Table 1).  The non-laying pairs each had a breeding territory and a nest-hole.  Reasons for failing to lay are unknown.

*Nest survival*

We previously estimated a nest survival rate of 51% (95% CI= 34 – 73%) at Nelson Lakes (Chapter 2). Accidents at the nest site caused 6% of these nest failures; eggs and chicks disappeared from the remaining 43%.

*Fledglings emerging from successful nests*

We determined the number of fledglings produced at 27 nests. Seven nests produced one chick, 17 produced two chicks and 3 produced three chicks (mean =1.85, se = 0.11). Productivity per successful nest was similar in all years (Kruskal-Wallis test, $\chi^2 = 3.733$, df = 4, p=0.445, n = 27 nests over 5 years).

*Fledgling sex ratio*

Of the eleven chicks whose gender was determined using molecular methods, six were male and five were female. Head size gave a clear indication of gender (Figure 2, n=13 chicks), so we used it to sex all of five chicks fledged from three nests in 1998. Two of these were male and three were female. The resulting sex ratio at fledging is one:one (total n = 18 chicks from eight nests).

**Figure 2. Gender determination and sex ratio of kea chicks at fledging. Head size is the only clear determinant of gender and the sex ratio at fledging is one: one.**
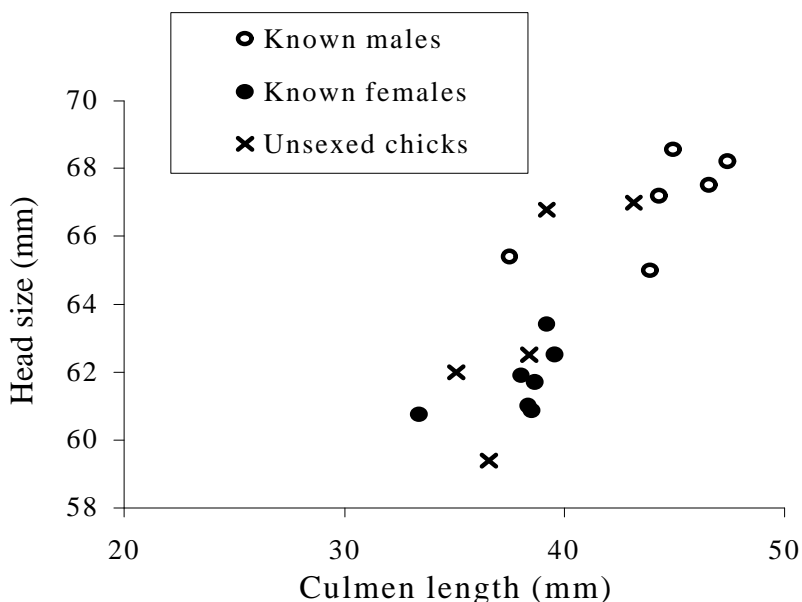
*Productivity per adult female per year*

The means of the parameters given above produce an estimated productivity of 0.33 (0.76 x 0.93 x 0.51 x 1.85/2) female chicks per adult female per year. This is probably a slight underestimate, as pairs will sometimes re-nest after early nest-failures.

**Adult survival**

*Somatic survival*

Three kea died in 1993 and two in 1996. All of the birds that died were male. One had puncture wounds in the head indicative of stoat predation or scavenging. Two were too decomposed to determine the cause of death. The fourth kea was seen vomiting in the skifield carpark two days before he was found dead, and the fifth drowned in a barrel full of water and rubbish. As two of the deaths seemed to have been caused by humans and we failed to determine cause of death in most cases, we did not test for differences between years. Pooling all years gives an overall adult survival rate of 93% (standard deviation = 2.8%).

*Adult female survival*

If 93% of adult females nest in breeding years and 43% of all nests are preyed upon (assuming that all disappearances of eggs or chicks were caused by predators), then their survival rate must be at least 53% (0.93-(0.93 x 0.43). This would be the survival rate if all nest failures resulted in the death of the female, but this does not appear to happen. The females from 5 of 11 failed nests had been marked with leg bands prior to the disappearance of their eggs/chicks and were searched for afterward. All five were confirmed alive after their nests were preyed upon. A mark-recapture type analysis of this data would suggest that females are seldom, if ever, killed by predators, in which case adult female survival equals somatic survival (93%). However, the sample size was small so we are not confident of this. If at least 45% (5/11) of adult females survive nest predation events, then the survival rate would be 71% (0.93 – (0.93 x 0.43 x 0.55). Adult female survival probably falls somewhere between 71% and the somatic survival estimate of 93%. Our best estimate is the midpoint of this interval, which is 82% (Table 1).
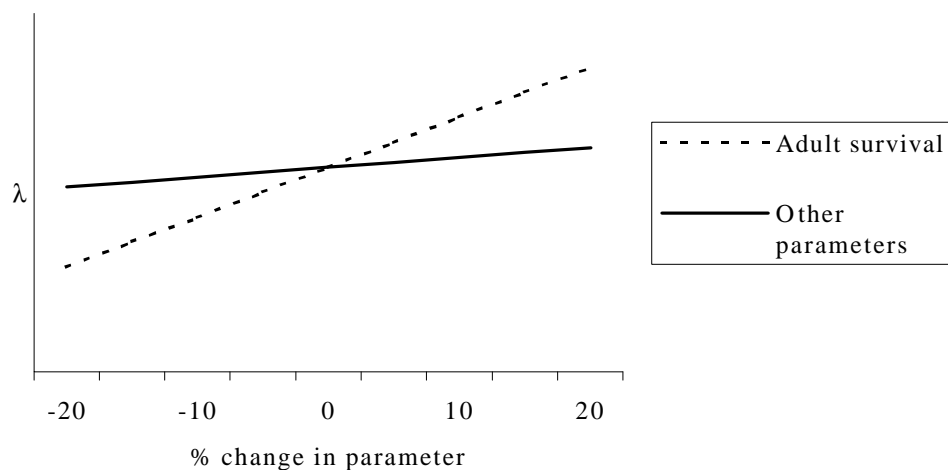
**Pre-adult survival**

Of the radio-tagged fledgling cohort, one fledgling died, one dropped its transmitter, one signal was lost and the remaining eight survived. Fledgling survival for 1998 is then 89% (95% CI = 70-100%, Table 1). Fledgling survival in 1998 may have been higher than average due to a particularly mild winter that year. We assume that juvenile survival is, on average, somewhere between fledgling survival and adult survival. Our estimates of average survival for fledglings, juveniles and sub-adults are 60%, 75% and 93% respectively.

**Population Dynamics**

Given the parameter estimates presented so far, $\lambda$ is over four times as sensitive to a change in adult survival as it is to an equivalent change in any other parameter (Table 1, Figure 4). All of the other parameters make equivalent contributions to $\lambda$.

**Figure 3. A graphical representation of elasticity. Changes in adult survival have a greater impact on $\lambda$ than proportional changes in the other parameters because adult survival makes a greater relative contribution to $\lambda$.**

## Discussion

### Adult survival

Our estimate of 93% annual adult survival (excluding nesting mortality) came from a small sample of individuals. It was, however, based on six years of data and variation during this period was low, so we are confident that the true value is high and stable. Including the two deaths from human-related causes may have resulted in a slight underestimation of the natural survival rate. Jackson (1969) also found a high survival rate at Arthur's Pass, but unfortunately he did not present separate survival estimates for the different age classes. His estimate for all ages combined was 63.3%. He noted that fledgling and juvenile kea were much more prone to death than adults, so his adult survival rate may have been similar to ours.

### Productivity

The number of chicks fledged from successful nests in our study (1.85) was similar to Jackson's (1.60) (1963). Kea in both study areas appear to be producing at a level below their physiological capabilities. Captive kea regularly produce 3 or 4 chicks per nest (Tony Pullar, New Zealand Captive Kea Co-ordinator, pers. comm.) and three kea nests we observed at a lower altitude site (300-600m a.s.l.; not presented in this paper) produced 3, 3 and 4 chicks respectively. Reasons for this are unclear, but it could relate to food availability in the montane/alpine habitat.

### Non-breeding years

The mechanisms involved in preventing breeding in 1996 and possibly 1991 are unclear. To investigate possible causes of the non-breeding years we looked at weather data gathered by the Department of Conservation at St Arnaud for the months of April to September for 1991 to 1998. We failed to find any obvious differences in weather patterns, all years being generally similar in rainfall, daily temperatures, frequency of frosts and severity of frosts. In contrast to our study, Jackson (Jackson 1969) observed seven consecutive breeding seasons in Arthur's Pass, although one was a dismal failure due to bad weather.

**Kea population dynamics**

We chose not to use more sophisticated stochastic population models, such as the population viability analysis techniques of Gilpin & Soule (1986), because such models would add little to our understanding of kea population dynamics given the available parameter estimates.  The continuity of kea habitat in South Island combined with our preliminary dispersal data (as yet unpublished) indicates that kea probably form one large population(except, perhaps, for kea in the Kaikoura Mountains).  As such, the kea population is too large (1000-5000 individuals, Anderson 1986), and kea live too long, for their extinction risk to be affected by demographic stochasticity.  In addition, our estimates of environmental variation in parameters are very limited.  The main utility of population models based on limited data sets lies in sensitivity analysis (Hamilton & Moller 1995; McCarthy *et al.* 1995) and the results of our analytical sensitivity analysis would be similar to results of sensitivity analyses using stochastic models.  Actually calculating the population growth rate and building more complex models should be left until more data is available.

The dynamics of our study population are driven largely by adult survival.  As long as adult survival is relatively constant, population trends will be relatively constant. A decrease in the adult survival rate will cause a much more rapid decline than a proportionally similar decrease in any other parameter.  Any sudden declines in kea numbers would, therefore, most likely be caused by inflated adult mortality.  We thought that kea may be prone to inflated adult mortality through predation because they spend a lot of time on the ground (nesting, feeding and roosting; personal observations) where they are more likely to encounter predators than if they were in the canopy all of the time.  However, we found that somatic survival was high (93%), the nest predation rate was relatively low and adult females seemed to escape being killed during the few nest predation events that we observed.  We therefore conclude that introduced mammalian predators are probably not causing a population decline among kea in our study area.  The nesting mortality sample was small, however, so we are cautious about placing too much weight on this result. We recommend that a larger sample of adult females be radio-tagged because the population growth rate is so sensitive to their survival.  If

nesting among a larger sample is also low, then the kea population is unlikely to require immediate conservation management and attention should turn to devising an effective monitoring method.

**Future Research**

We have recommended that future research should focus mainly on verifying our estimate of adult female survival.  We also need to improve on our estimates of pre-adult survival, and measure breeding frequency, the proportion of adults that breed and productivity among kea living in the hinterlands. The development of radio-transmitters with five years of battery life enables the concurrent implementation of these recommendations.  Our suggestion is to continue radio-tagging fledglings in nests in our study site.  Most of these fledglings should disperse to areas away from the skifield (this study unpublished data).  Pre-adult survival and dispersal data can be collected each year as these fledglings mature and by the end of five years battery life they will probably have settled on breeding territories, thereafter forming a sample of radio-tagged adult females sustained only by natural food.  Replacing the transmitters on the females and radio-tagging the mates of the males should result in a sample of 15-20 adult females by the year 2003. Once mature, these birds can be visited in mid-spring to see if they are breeding and, if so, again in late summer to see if they survived.  The relative age of first breeding between males and females should be indicative of the sex ratio within the kea population.  Females' beginning to breed at a younger age than males would suggest that the sex ratio is skewed toward males.  Such a skew would most likely result from predation on nesting adult females (Wilson *et al.* 1998), so the amount of skew should provide a second indication of the amount of nesting mortality that is occurring. Research aimed at achieving all of these goals is underway at Nelson Lakes. Eleven fledglings were radio tagged in 1997 and seven were tagged in 1998.   Funding is secured (from the Department of Conservation) to track these two cohorts this summer and to radio-tag a third cohort of nine fledglings.

The results from the skifield study indicate that kea are unlikely to require immediate management action.  This may be a rare case for New Zealand where further research

could result in a downgrading of the conservation status of the species in question. Therefore, rather than taking our results as a reason to stop research into kea population dynamics, we take them as a strong reason to continue with the recommended further research.

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# 4.  Research summary and recommendations

Given the results presented in the preceeding chapters, predators appear to be having only a minor impact on kea abundance.  This suggests that the current low density of kea in the Southern Alps either reflects a return to a pre-European state or that kea numbers are slowly declining through more subtle mechanisms such as competition for food, possibly combined with a low level of nest predation.

As we can not be certain that similar results would be found elsewhere in the Southern Alps, similar studies in different parts of South Island would be an invaluable complement to this study.  These would best be aimed at kea who feed only on natural food, as most kea fall into this category.  If levels of nest predation and adult survival in other sites are similar to what we have observed in Nelson Lakes, then a high degree of confidence could be placed in a decision not to implement management specifically for kea conservation.

In deciding whether or not to replicate this study, managers should remember that short term studies of species like kea are to be avoided.  We anticipate a low capture rate (for radio-tagging) among kea living away from skifields because they dod not recognise humans as a source of food and so tend to keep to themselves (pers. obs.).  With perseverance, however, the yearly expense of such a study should decrease and sample sizes will increase, as more transmitters are put out into the field and as more nest holes become known and are re-used.  Established study areas with known nest holes will be useful long into the future.  We would have returned to Jackson's nest holes at Arthur's Pass had records of their location been kept.

Future research should focus mainly on verifying our estimate of adult female survival. We also need to improve on our estimates of pre-adult survival, and measure productivity among kea that do not scavenge food from humans.  The development of radio-transmitters with five years of battery life enables the concurrent implementation of these recommendations.  Our suggestion is to radio-tag a third cohort of fledglings produced in

the Nelson Lakes study site. Most of these fledglings should disperse to areas away from the skifield (this study, unpublished data). Pre-adult survival and dispersal data can be collected each year as these fledglings mature, and by the end of five years they will probably have settled on breeding territories, thereafter forming a sample of radio-tagged adults who are sustained only by natural food. Replacing the transmitters on the females and radio-tagging the mates of the males should result in a sample of 15-20 adult females by the year 2003. Once mature, these birds can be visited in mid-spring to see if they are breeding and, if so, again in late summer to see if they survived.

Research aimed at achieving all of these goals is underway at Nelson Lakes. Eleven fledglings were radio-tagged in 1997 and seven were tagged in 1998. Funding is secured (from the Department of Conservation) to track these two cohorts this summer and to radio-tag a third cohort of nine fledglings.

The relative age of first breeding between males and females in the radio-tagged fledgling cohorts should be indicative of the sex ratio within the kea population. Females' beginning to breed at a younger age then males would suggest that the sex ratio is skewed toward males. Such a skew should provide a second indication of the amount of nesting mortality that is occurring in the population.

A summary of our recommendations is as follows-

1) The locations of 29 kea nest holes are known. As nest holes are often re-used, this provides a unique opportunity to quickly find nests without the use of radio-telemetry. We recommend that each of the 29 nest holes be visited yearly in September. If they contain nests, they should be re-visited in December to see if they have been successful. At the very least, this should happen in the breeding seasons following heavy beech seeding when stoat numbers tend to increase.

2) An attempt should be made to colour-band or radio-tag the adult females associated with these nests so that their fate after nest failure can be determined. The number of colour-banded adult females in the study area stood at seven at the end of the 98/99 field season and the number of radio-tagged adult females stood at one.

3) The fledgling cohorts described above should be located and visited each summer for the next five years to improve knowledge of age-specific mortality and dispersal. Ideas for a survey method may come to hand in this process.

4) When these cohorts mature, radio-transmitters should be replaced on the females and fitted to the mates of the males to produce a sample of 20 radio-tagged adult females in the hinterlands. Adult female survival and productivity in the hinterlands can then be estimated from these birds. We anticipate that this will involve 8-12 person-weeks of work per year, at a cost of less than $10 000 per year.

This may be a rare case for New Zealand where further research could result in a downgrading of the conservation status of the species in question. Therefore, rather than taking our results as a reason to stop research into the conservation ecology of kea, we take them as a strong reason to continue with the recommended further research.