




Influence of human-wildlife conflict on the population dynamics of Kea (*Nestor notabilis*)


E.M. Kennedy *Toi-Ohomai Institute of Technology*
C.E. Simpkins *University of Göttingen, University of Auckland*
G.L.W. Perry *University of Auckland*

This is an abstract

Keywords: These are keywords


Introduction

Humans are rapidly expanding into **wildlife areas**, causing an increase in human-induced mortality (**HIM**). Human-wildlife conflict (HWC) is widely reported because it affects humans ~~as well as~~ wildlife populations ~~themselves~~ (e.g., wildlife attacks on humans, crop raiding, and property destruction Woodroffe, Thirgood and Rabinowitz (2005)). A number of conflict-prone species are threatened or endangered (e.g., African elephants or Sumatran tigers), and it is assumed that HIM plays a role in causing and/or maintaining population declines (Thouless, 1994; Landa et al., 1999). However, it is important that managers ascertain the magnitude of the impact of HIM on a specific population, as well as considering the relative influence of HIM compared to other factors. Managers often have limited resources (e.g. money, time, and equipment) and need ~~detailed~~ information about whether it might be more practical to focus on mitigating all threats equally, or, alternatively, focus more intensively on a smaller suite of particular threats/limiting factors. Despite this need to measure the impacts of HIM, studies explicitly quantifying these impacts are rare  this shortfall may be due to many conflict scenarios being relatively new (and constantly changing), so demographic data capturing the effects of human-wildlife conflict are scarce.

Kea are one of New Zealand's best examples of a species that encounters HWC. They are the world's only mountain- and rainforest-dwelling parrot (Greer, Gajdon and Nelson, 2015), and have an innate intelligence and curiosity that stems from the need to source a wide variety of food (Diamond and Bond, 1999; Auersperg et al., 2011). Historically, this curiosity led to conflict after high-country farmers reported incidences of kea attacking sheep to eat fat from around the kidneys, **often** leading to death of the sheep from sepsis (Orr-Walker and Roberts, 2009). Consequently a bounty was instituted by the government to cull kea, leading to c. 150,000 individuals being killed over a 100-year period (Temple, 1996), before the species was afforded full protection under the Wildlife Act in 1986. Kea damage human property, and there are still reports of kea strike on sheep, resulting in ~~both~~ direct persecution and indirect human-induced mortality (Reid, pers. comm.). Even attempts to protect the kea have created unexpected HIM; 1080 (sodium fluoroacetate), which has been used as a poisoning agent to control introduced mammalian predators, has been found to cause the accidental death of some adult kea through ingestion of the bait (Orr-Walker et al., 2012)  Due to these **on going** human threats (Gartrell and Reid, 2007), ~~as well as~~ pressure from mammalian predators (Kemp et al., 2018), kea populations continue to be at risk (Elliott and Kemp, 2004). Recently, kea were classified as 'Nationally Endangered' by the New Zealand Department of Conservation **[DOC2016]** and ~~currently~~ the IUCN classification is 'Endangered' (IUCN, 2017). Despite the kea's conservation status, little is known about the influence of HIM on kea population dynamics, or how the likely growth of the tourist population in kea

habitat areas will alter these dynamics.



Population viability analysis (PVA) is an effective means of quantifying population dynamics, and can provide essential information for management of threatened and endangered species (Boyce, 1992; Morris and Doak, 2002). PVAs are quantitative models informed by demographic data, which allow researchers to evaluate and predict how ~~different~~ biotic and abiotic factors will affect population growth or decline over time (Beissinger and Westphal, 1998; Mills and Lindberg, 2002). Historically, PVAs have been used primarily as predictive tools, e.g. to calculate minimum viable populations and to predict absolute values of future populations (Boyce, 1992; Ginzburg et al., 1982). However, as models are simplifications of the systems and phenomena they seek to represent there is always uncertainty associated with their predictions (Coulson et al., 2001; Ellner et al., 2002). More recently, it has been recommended that PVAs are used more to explore the qualitative **differences in systems** rather than making predictions based on outputs (Simpkins et al., 2018), and this has been shown to be highly effective in understanding the relative importance of different factors in driving population dynamics (e.g. Simpkins et al., 2015). A number of studies have used the PVA framework to explore the relative impact of HIM on conflict prone species [e.g LaFever et al. (2008); **goswami2014**], ~~however this number is still limited~~. By understanding how HIM and other threats influence population dynamics, the relative importance of these factors, and how their influence may vary over time, managers are better able to develop effective management strategies.

Our goal ~~in this paper~~ was to explore how changes in the human population, in kea habitats, would impact kea population dynamics, and thus determine what, ~~if any, additional~~ management steps are needed to ensure the long term survival of the species. In order to achieve this goal we developed an age-structured, density-dependent model of kea  We used this model to investigate how possible alternative types of human population growth, with resultant increases in HIM, could impact the long term viability of kea.

Materials and Methods

Study Specie

The kea is a large, omnivorous parrot (family **Strigopidae**) restricted to the South Island of New Zealand (Figure 1). ~~Habitat varies~~ from coastal dunes to high alpine peaks but ~~kea~~ are most common in high **altitude** southern beech (*Nothofagaceae*) forest, sub-alpine shrublands, and high-alpine basins and ridges (Higgins, 1999; Robertson, 2007). The current kea population size is uncertain, but the most recent estimate ~~of overall population size gives numbers of~~ between 1000-5000 wild birds (Anderson, 1986). It is difficult to precisely estimate kea numbers due to their extensive range (largely in rugged terrain), low density, and the cryptic behaviour of adults (Orr-Walker and Roberts, 2009).

The maximum life span of kea in the wild is thought to be c. 25 years, but birds in captivity have ~~been shown to live~~ more than 47 years (Brouwer et al., 2000). Kea are non-territorial, and form monogamous long-term pairs (Bond, Wilson and Diamond, 1991). They nest on the ground in crevices, usually below the treeline (McCaskill, 1954). Females generally become sexually mature between 3 and 4 years of age (Jackson, 1963). Individuals nest between July and  uary, with between 1 and 5 eggs being produced. Incubation takes 22-24 days, and chicks fledge  in approximately 90 days. Kea chicks have a long juvenile period and are dependent on their parents for 4-5 months after hatching (Orr-Walker, 2010).

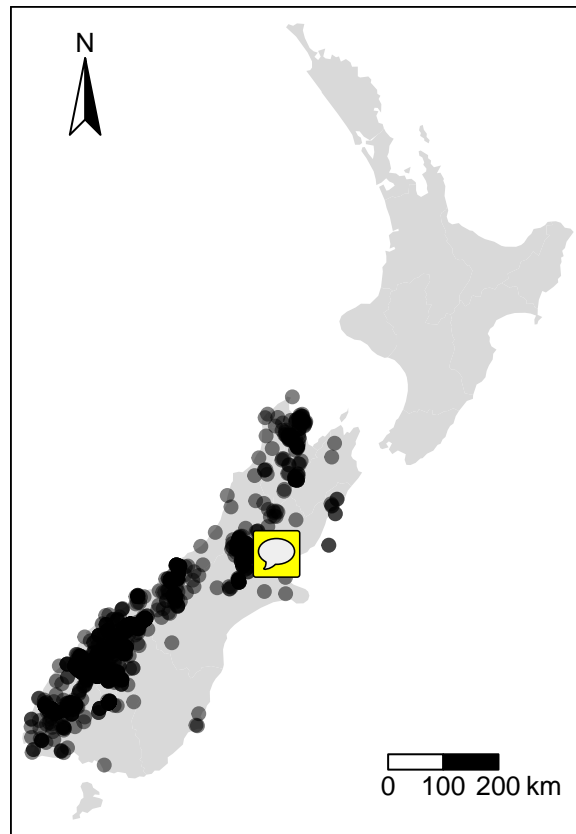


Figure 1: Kea **sittings** recorded since 1995 in the South Island of New Zealand (data taken from GBIF (2020))

Model Structure

We developed a stochastic simulation model based around three age classes to explore the influence of HIM on kea population dynamics. The three age classes were juveniles (0-1 year), sub-adults (1-3 years), and adults (3+ years). These age classes were selected to match significant changes in behaviour, mortality risk, and breeding ability in the life history of the species. The model simulated the transition between each age class, breeding, and mortality (Figure 2). Mortality rates were made up of background and predation mortality along with different levels of HIM based on the scenario being explored.

Only females were represented in the model as the kea's monogamous reproductive status indicates that unpaired males rarely contribute to population growth (Bond, Wilson and Diamond, 1991; Ferson and Burgman, 1995). Furthermore, previous research suggests there is a male sex-bias in wild populations, meaning that females are more likely to be able to form pair-bonds than males (Bond, Wilson and Diamond, 1991). Due to the high degree of uncertainty the statistical associations between demographic parameters in the model were perfectly correlated; although this correlation in vital rates is unlikely, it provides a more conservative approach to assessing the population's viability (Ellner et al., 2002; Ferson and Burgman, 1995).

The model was constructed in R v3.6.2 (R Core Team, 2019), using the deSolve package (Soetaert, Petzoldt and Setzer, 2010). The model ran for 100 timesteps, simulating 100 years (each timestep representing one year). This period was chosen as it was long enough to detect demographic trends in the long-lived species, while still being a relevant timeframe from a management perspective.

Model parameters

Population parameters

The initial population size of kea was based on the lowest published estimate of 500 females (Pullar, 1996). We used the lowest value because there is considerable uncertainty in population estimates (Orr-Walker and Roberts, 2009), and we wanted to present a conservative rather than optimistic estimate. We assumed a stable age-distribution for the initial timestep.

Reproduction

Kea have a mostly monogamous breeding system (Bond, Wilson and Diamond, 1991), and are reported to begin breeding from three years of age (Jackson, 1963). To simulate this behaviour only adult birds (over 3 years of age) were capable of breeding in our model. Fecundity was defined as the product of the number of adult females, clutch size, and egg survival. The proportion of breeding adults was set at 0.6 as not all females breed successfully every year (Bond, Wilson and Diamond, 1991). Clutch size for kea was generated using a Poisson distribution with a rate (λ) of 1.35. This value was selected to match the observed kea clutch size of between 1-5, taking into account that only approximately half the chicks would be female (Bond and Diamond, 1992).

Mortality

Mortality rates varied between each of the three age classes (Table 1). Baseline mortality rates were estimated as a combination of both natural events (e.g. illness) and predation pressure under the current introduced management program (DOC, 2007). In addition, approximately every four years a beech mast occurs (Ogden, Stewart and Allen, 1996), triggering irruptions of mammalian predators which often predate kea, leading to an increase in predation mortality (Elliott, 1996;

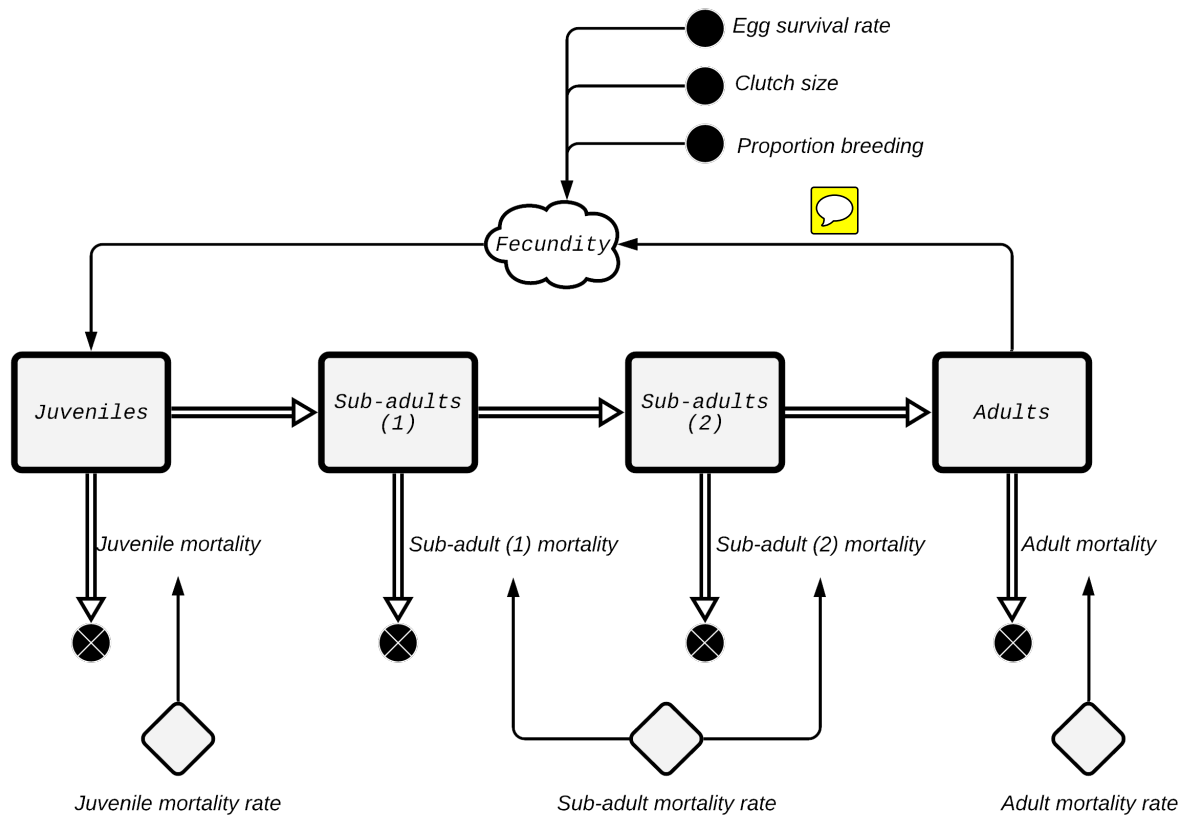


Figure 2: Overview of the kea population model. Rectangular boxes denote the population stocks for each age class, note there are two sub-adult stocks to denote that individuals remain in this class for two timesteps. The large double-lined arrows represent the flow of individuals. The single-lined arrows represent connections between model parameters.

Choquenot, 2006). This was accounted for by adding a mast event with a 0.25 probability of occurring each time step. During a mast event the predation rate in the system increased by for that time step. Stochasticity was added to the baseline mortality rates by multiplying them by values drawn from a random uniform distribution with values 10% above and below the standard rate.

Table 1: Initial demographic values used in the PVA model.
N is the initial number of individuals. Rates are all expressed per timestep/year.

Age class	N	Mean baseline mortality rate	Current HIM rate
Egg	-	0.4	0
Juvenile	125	0.2	0.01
Sub-adult	250	0.05	0.015
Adult	125	0.1	0.015

Department of conservation's database was used to determine how many kea deaths could be attributed to HIM. We counted intentional causes (e.g. shooting, trapping) and accidental causes (e.g. vehicle strike) as HIM. While predation by introduced mammals could be considered a human induced effect, it is a ~~large~~, well studied, and heavily managed factor which we argue is beyond the scope of other HIM types. HIM rates were the same for all age-classes, though no HIM was assigned to eggs, as kea nests are cryptic and there is minimal chance of people interacting with them. To represent potential increases in HIM due to human population growth over time we evaluated four potential population growth types: 1) No growth (baseline); 2) linear; 3) exponential; and 4) logistic.

Linear growth was calculated as:

$$Mort_{t+1} = Mort_t + C$$

Where C is the rate of change in the population.

Exponential growth was derived using:

$$Mort_{t+1} = Mort_t(1 + C)^t$$

Where t is the timestep and C is the rate of change in population.

Logistic growth was represented using:

$$Mort_{t+1} = C.Mort_t\left(\frac{1 - Mort_t}{1}\right)$$

Where t is the timestep and C is the rate of change in population.

Scenarios tested

We evaluated four ~~different~~ scenarios representing different potential human population growth scenarios (Table 2.)

Table 2: Description of the scenarios tested using the PVA; Growth type is the type of human population/tourist population growth used throughout a simulation; C is the constant for rate of growth **appropriately** used for that growth type.

Scenario	Growth type	C	Description
I	None	-	Baseline scenario in which current conditions are maintained. Used to identify the relative change in the other scenarios
II	Linear	0.005	Simulating a constant increase in population/tourism numbers and the resultant increase in HIM
III	Exponential	0.01	Simulating an uncontrolled exponential growth in population/tourism numbers
IV	Logistic	0.005	Simulating an increase in population/tourism numbers which has a slowed rate of increase as facilities in the area reach capacity

Model execution and analysis

Each scenario was evaluated by running 1000 simulations of 100 timesteps each (unless the population reached zero before this time). We used a critical population size as a ‘quasi-extinction’ threshold (Morris and Doak, 2002), as this is often used to set conservation policy (Mace and Lande, 1991). We defined the quasi-extinction threshold as ~~being a~~ population of 50 individuals (Holmes et al., 2007; Otway, Bradshaw and Harcourt, 2004). Time until extinction was determined by calculating the first year in which a quasi-extinction threshold was passed. The overall extinction risk was estimated as the proportion of replicates for each scenario which experienced a quasi-extinction (i.e. $N < 50$).

Additionally, mean population growth rates for each scenario were determined using the mean geometric growth:

$$\lambda = \left(\frac{N_t}{N_0} \right)^{\frac{1}{t}}$$

Where λ is the growth rate; N_t is the number of kea at time t ; and N_0 is the number of kea at time zero.

We also conducted a univariate sensitivity analysis for clutch size and the proportion of breeding adults, which were determined to be demographic values with a ~~relatively~~ high level of uncertainty. The model was ~~run~~ using baseline conditions (Scenario I ~~i.e. current conditions~~). Each parameter was individually varied by $\pm 10\%$ of their baseline value and the model run for 100 timesteps for 1000 repeats with each value. As the clutch size was drawn from a Poisson distribution the λ of this distribution was altered. The parameter sensitivities were determined using the index described by Hamby (1994).

$$S_{y,x} = \frac{\left(\frac{\Delta y}{y_b} \right)}{\left(\frac{\Delta x}{x_b} \right)}$$

Where $S_{y,x}$ is the sensitivity of the output variable y to a change in the input variable x . Δy is the change in output variable y , y_b is the baseline output. Δx is the change in input variable x , x_b is

the baseline input. The population size of kea was used as the output variable of interest and a parameter was considered to be 'sensitive' if the index resulted in a value > 1 .

Results

Of the 4000 ~~total~~ runs only 47 (1.175) resulted in extinction events (i.e. population dropping below 50 individuals). The exponential human growth scenario resulted in the most extinctions (22 runs) though this was still a small proportion of the 1000 repeats run for this scenario. The other three scenarios all showed ~~very~~ similar rates of extinction (linear = 9 runs; logistic = 5 runs; no growth = 11).

All four scenarios had average populations ~~well~~ below both carrying capacity (1000 individuals), or the initial population (500 individuals; Figure 3). The no growth scenario had a higher average population than any of the other three scenarios ($\lambda = 321.84 \pm \sigma = 49.99$; Hedge's $G > 0.8$ for all scenarios). The linear growth scenario has the lowest average population (269.68 ± 47.57), with this difference having a noticeable effect size between both the exponential growth scenario (Hedge's $G = 0.01$) and the logistic growth scenario (Hedge's $G = 0.27$). The exponential and logistic growth scenarios had very similar average populations (282.14 ± 47.77 and 282.66 ± 47.73 respectively; Hedges $G = -0.26$). These population differences between scenarios appear consistent through time (Figure 4). All scenarios showed a similar pattern through time with a steep decline followed by a long term stable population. Due to all the scenarios having the same general trend, with long stable periods the growth rate for all the scenarios were ~~very~~ similar (Hedge's $G < 0.1$; Figure 5), with all showing a slightly negative mean growth rate.

Sensitivity Analysis

~~Of the four sensitivity scenarios tested the~~ model was ~~found to~~ only be sensitive to a decrease in the proportion of breeding females (sensitivity index = 1.54). When the proportion of breeding females was lowered the average population decreased compared to the baseline (304.03 ± 47.53 ; 359.44 ± 52.87 respectively; Hedge's $G = 0.69$), meaning that if the proportion of females is lower than our estimate our results may be optimistic.

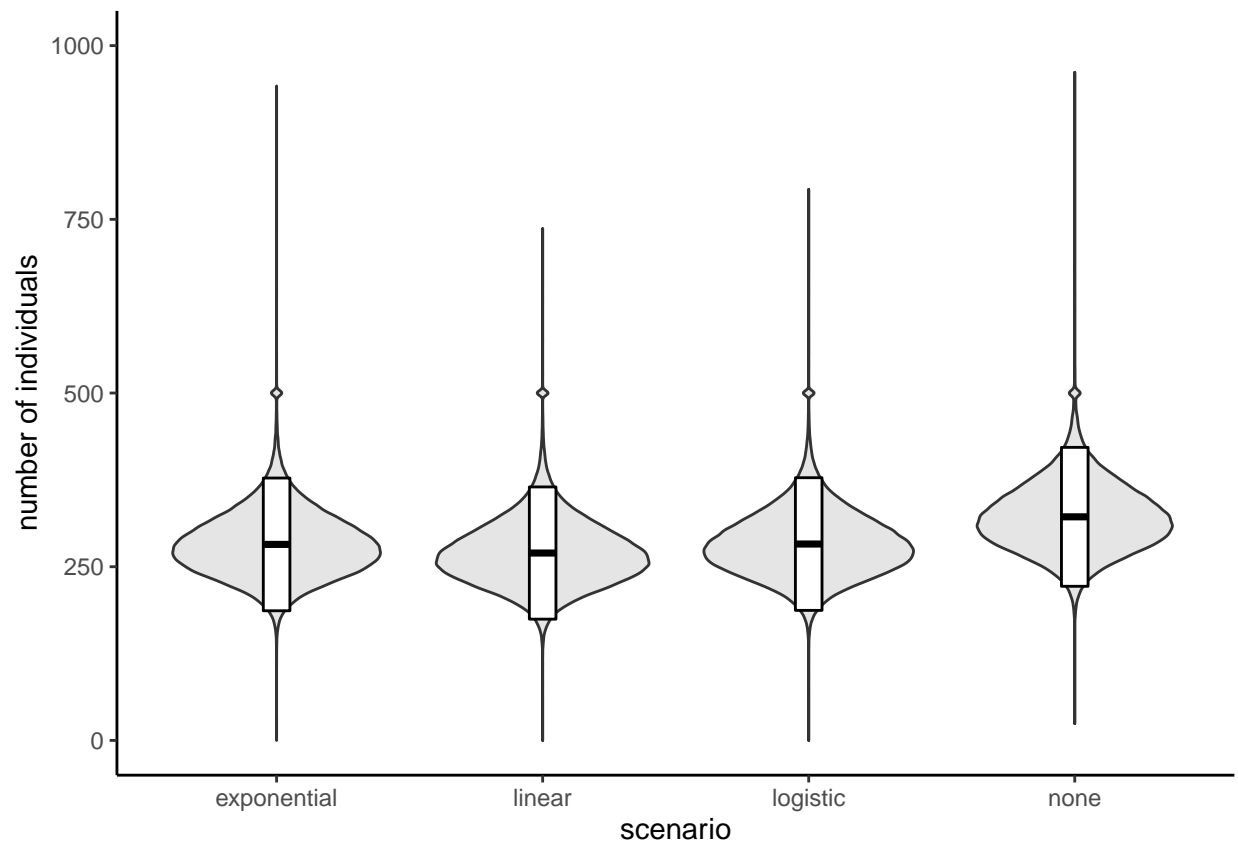


Figure 3: Mean number of individuals over the 250 time step period for all 1000 simulations for each scenario.

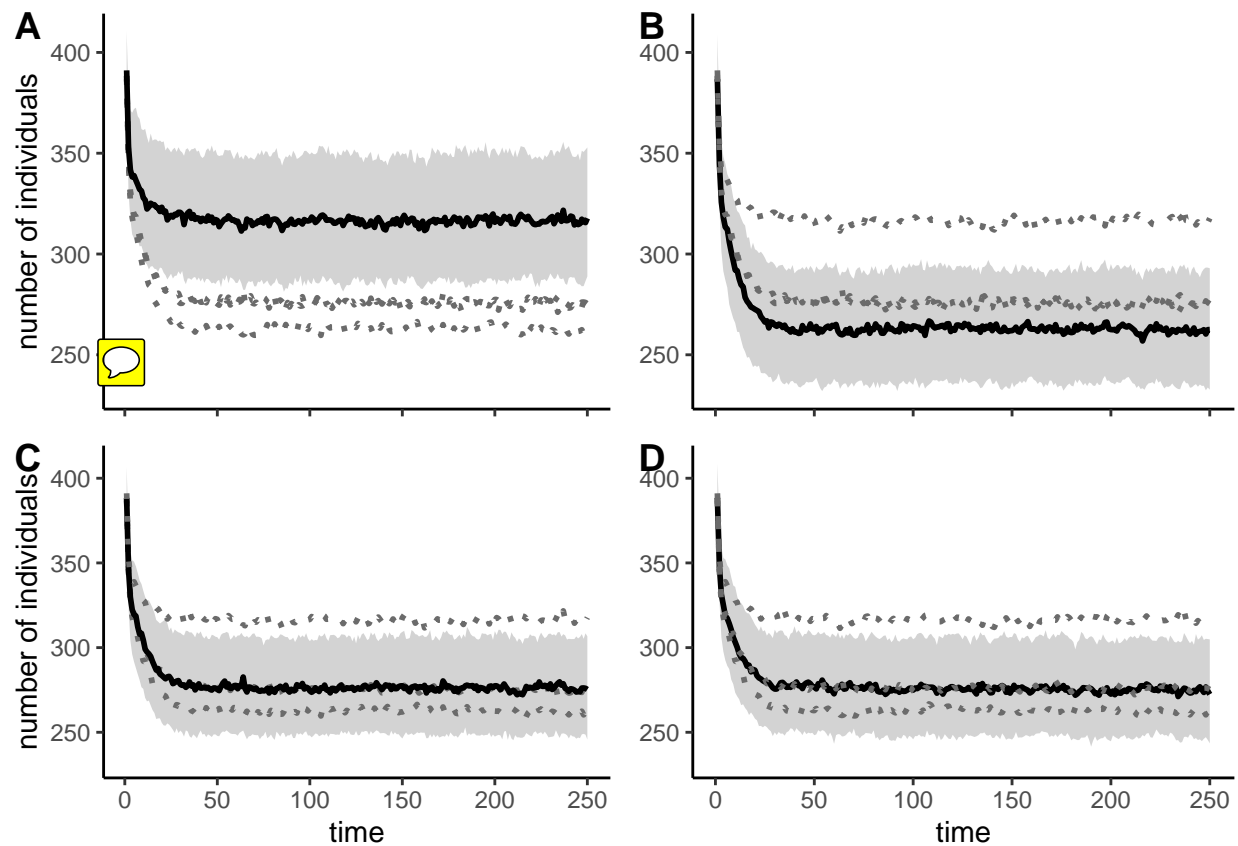


Figure 4: Change in the number of individuals through time for each of the tested human population growth scenarios. A is the no growth scenario; B is the linear growth scenario; C is the logistic growth scenario; D is the exponential growth scenario. The solid black line represents the mean number of individuals over the 1000 repeats for each scenario, with the gray bands representing one standard deviation above and below this mean. The dotted gray lines represent the means for the other three scenarios.

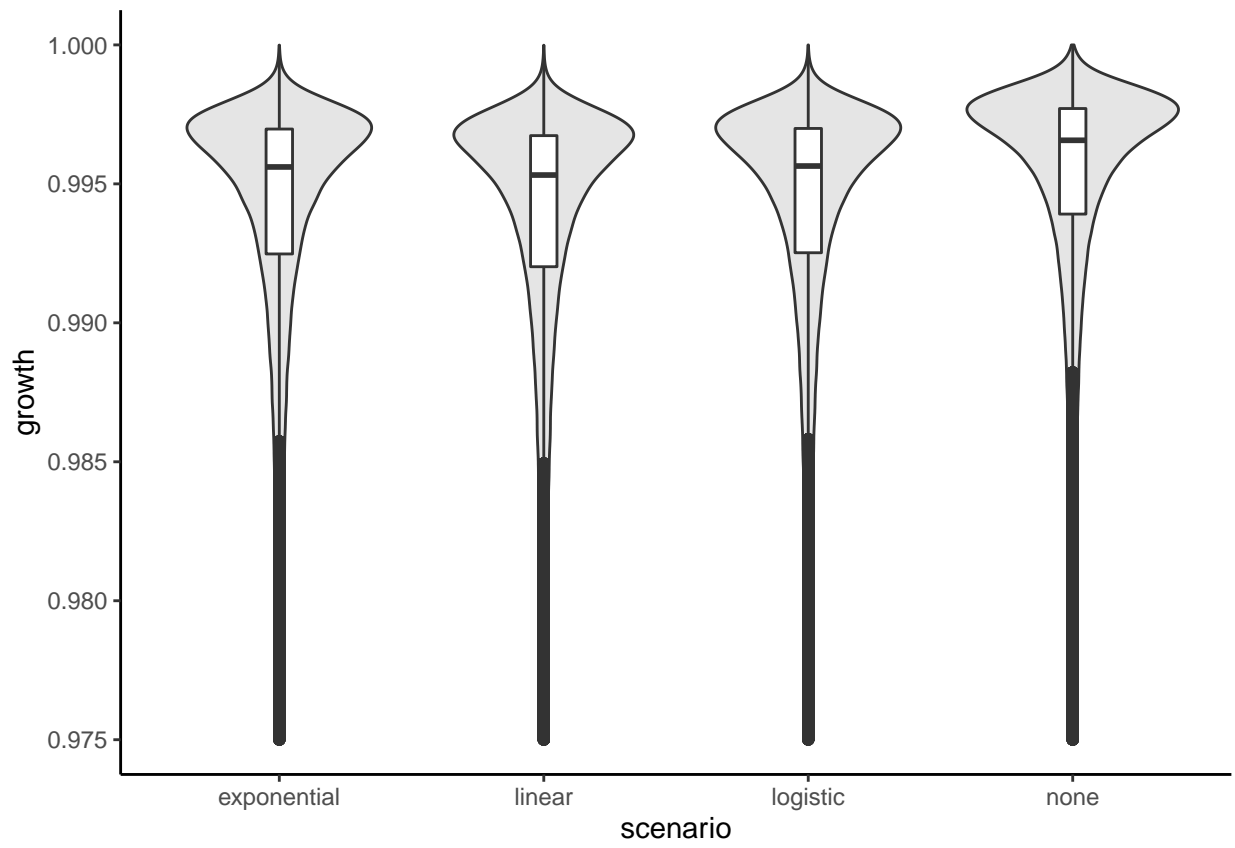


Figure 5: Mean growth over the 250 time step period for all 1000 simulations for each scenario.

Discussion

Our study found that the kea population has a low chance of extinction if conditions remain stable. However, under all the scenarios tested the population did show a gradual decline. Therefore, the population will become increasingly more sensitive to the effects of disturbances or external factors not included in our model, such as climate change (Keith et al., 2008). While very few estimates of kea population trends have previously been made, those that have found that the kea population was likely stable (Bond and Diamond, 1992). This discrepancy between historical projections and our current analysis may indicate a worrying change in the population's trajectory, indicating that pressures beyond those being managed for are negatively impacting the population. Therefore, additional pressures, such as those imposed by increasing human conflict may further tip the species into decline.

The kea population was found to be very resilient to changes in HIM. While differences were found in the overall number of extinctions and rate of population decline between the different human population growth scenarios, these differences were minor and did not alter the overall observed kea population trend. It is likely that the kea population may be even less sensitive to changes in human population than what we predict. This is because while we predict uniform changes in the human population, i.e. both urban and rural populations will show equal growth, it is predicted that future growth will be primarily in urban areas, away from the kea's alpine habitats. Though it should be noted that if urban areas begin to sprawl into kea habitat areas this effect will obviously be reversed. Due to this likely resilience, HIM can be seen to not be a currently significant issue in kea conservation, especially as kea have been found to be heavily impacted by introduced mammalian predators, with their fecundity declining greatly in non predator managed areas (Kemp et al., 2018). However, while it is likely not necessary to manage explicitly for HIM, it should none the less be considered at the planning stage, and when making population projections, as it may act to reduce the population sufficiently to tip it below a sustainable size. The management of some HIM may also be seen as low hanging fruit, requiring little effort to maintain or decrease human-wildlife conflict events, and as such, may be a cost effective parameter to manage.

While our results show that the kea population is resilient to changes in HIM, it should be noted that a number of factors may reduce this resilience. Firstly, it is likely HIM rates are higher than were included in our model, this is because many HIMs may not be reported or detected, though this increase is difficult to quantify. Secondly, we did not include indirect or secondary HIM events, as these are difficult to assign as being human-induced events. Of particular concern is the changes in mortality rates that may occur due to kea ingesting human food, such as chocolate, which may have long term detrimental health effects (Gartrell and Reid, 2007). Though, clearly the access to additional resources in times of scarcity may also decrease mortality rates during some periods. Secondly, we found that our model was sensitive to decreases in the proportion of breeding females. As a result if the proportion of breeding females was below our estimate or it decreases over time the population will become more sensitive to HIM, amplifying any negative effects resulting from an increase in the human population. Due to the factors it is important that the results of our model be viewed qualitatively, as opposed to quantitatively, a recommendation made for most population models (Coulson et al., 2001). These uncertainties also help highlight our recommendation that, while HIM may not be the most pressing issue on kea populations, it needs to be included as an important factor in future management planning.

Acknowledgements

References

- Anderson, R. 1986. "Keas for keeps." *Forest and Bird* 17(1):2–5.
- Auersperg, Alice MI, Auguste MP Von Bayern, Gyula K Gajdon, Ludwig Huber and Alex Kacelnik. 2011. "Flexibility in problem solving and tool use of kea and New Caledonian crows in a multi access box paradigm." *PLoS One* 6(6).
- Beissinger, Steven R and M Ian Westphal. 1998. "On the use of demographic models of population viability in endangered species management." *The Journal of wildlife management* pp. 821–841.
- Bond, A.B., K.J. Wilson and J. Diamond. 1991. "Sexual dimorphism in the kea *Nestor notabilis*." *Emu* 91(1):12–19.
- Bond, Alan B and Judy Diamond. 1992. "Population estimates of kea in Arthur's Pass National Park." *Papers in Behavior and Biological Sciences* p. 31.
- Boyce, M.S. 1992. "Population Viability Analysis." *Annual Review of Ecology and Systematics* 23(1):481–497.
URL: <https://doi.org/10.1146/annurev.es.23.110192.002405>
- Brouwer, K., M.L. Jones, C.E. King and H. Schifter. 2000. "Longevity records for Psittaciformes in captivity." *International Zoo Yearbook* 37(1):299–316.
- Choquenot, David. 2006. "Bioeconomic modeling in conservation pest management: Effect of stoat control on extinction risk of an indigenous New Zealand passerine, *Mohua ochrocephala*." *Conservation Biology* 20(2):480–489.
- Coulson, Tim, Georgina M Mace, Elodie Hudson and Hugh Possingham. 2001. "The use and abuse of population viability analysis." *Trends in Ecology & Evolution* 16(5):219–221.
- Diamond, Judy and Alan B Bond. 1999. *Kea, bird of paradox: the evolution and behavior of a New Zealand parrot*. Univ of California Press.
- DOC. 2007. Arthur's Pass National Park management plan. Technical report Department of Conservation.
- Elliott, G and J Kemp. 2004. "Effect of hunting and predation on kea, and a method of monitoring kea populations: results of kea research on the St Arnaud Range. DOC Science Internal Series 181. Department of Conservation, Wellington, New Zealand."
- Elliott, Graeme P. 1996. "Mohua and stoats: a population viability analysis." *New Zealand Journal of Zoology* 23(3):239–247.
- Ellner, Stephen P., John Fieberg, Donald Ludwig and Chris Wilcox. 2002. "Precision of Population Viability Analysis." *Conservation Biology* 16(1):258–261.
URL: <https://conbio.onlinelibrary.wiley.com/doi/abs/10.1046/j.1523-1739.2002.00553.x>
- Ferson, Scott and Mark A. Burgman. 1995. "Correlations, dependency bounds and extinction risks." *Biological Conservation* 73(2):101 – 105. Applications of Population Viability Analysis to Biodiversity Conservation.
URL: <http://www.sciencedirect.com/science/article/pii/0006320795900314>

- Gartrell, BD and Clio Reid. 2007. "Death by chocolate: a fatal problem for an inquisitive wild parrot." *New Zealand Veterinary Journal* 55(3):149–151.
- GBIF. 2020. "*Nestor notabilis* Occurrence."
- Ginzburg, Lev R, Lawrence B Slobodkin, Keith Johnson and Andrew G Bindman. 1982. "Quasiextinction probabilities as a measure of impact on population growth." *Risk Analysis* 2(3):171–181.
- Greer, Amanda L, Gyula K Gajdon and Ximena J Nelson. 2015. "Intraspecific variation in the foraging ecology of kea, the world's only mountain-and rainforest-dwelling parrot." *New Zealand Journal of Ecology* 39(2):254–261.
- Hamby, DM. 1994. "A review of techniques for parameter sensitivity analysis of environmental models." *Environmental monitoring and assessment* 32(2):135–154.
- Higgins, P.J. 1999. *Handbook of Australian, New Zealand & Antarctic Birds. Vol. 4, Parrots to Dollarbird*. Oxford University Press.
- Holmes, Elizabeth Eli, John L Sabo, Steven Vincent Viscido and William Fredric Fagan. 2007. "A statistical approach to quasi-extinction forecasting." *Ecology Letters* 10(12):1182–1198.
- IUCN. 2017. "The IUCN Red List of Threatened Species 2017.".
URL: <https://dx.doi.org/10.2305/IUCN.UK.2017-3.RLTS.T22684831A119243358.en>
- Jackson, J.R. 1963. "The nesting of keas." *Notornis* 10(8).
- Keith, David A, H. Resit Akçakaya, Wilfried Thuiller, Guy F Midgley, Richard G Pearson, Steven J Phillips, Helen M Regan, Miguel B Araújo and Tony G Rebelo. 2008. "Predicting extinction risks under climate change: coupling stochastic population models with dynamic bioclimatic habitat models." *Biology Letters* 4(5):560–563.
URL: <https://royalsocietypublishing.org/doi/abs/10.1098/rsbl.2008.0049>
- Kemp, Joshua R, Corey C Mosen, Graeme P Elliott and Christine M Hunter. 2018. "Effects of the aerial application of 1080 to control pest mammals on kea reproductive success." *New Zealand Journal of Ecology* 42(2):158–168.
- LaFever, David H, Paige M Schmidt, Neil D Perry, Craig A Faulhaber, Roel R Lopez, Nova J Silvy and Elizabeth A Forsys. 2008. "Use of a population viability analysis to evaluate human-induced impacts and mitigation for the endangered Lower Keys marsh rabbit." *Human-Wildlife Conflicts* 2(2):260–269.
- Landa, A., K. Gudvangen, J.E. Swenson and E. Røskft. 1999. "Factors associated with wolverine *Gulo gulo* predation on domestic sheep." *Journal of Applied Ecology* 36(6):963–973.
URL: <https://besjournals.onlinelibrary.wiley.com/doi/abs/10.1046/j.1365-2664.1999.00451.x>
- Mace, Georgina M and Russell Lande. 1991. "Assessing extinction threats: toward a reevaluation of IUCN threatened species categories." *Conservation biology* 5(2):148–157.
- McCaskill, M. 1954. "The Poutini Coast: a geography of Maori settlement in Westland." *New Zealand Geographer* 10(2):134–150.
- Mills, L.S. and M.S. Lindberg. 2002. *Population viability analysis*. University of Chicago Press chapter Sensitivity analysis to evaluate the consequences of conservation actions., pp. 338–366.

- Morris, William F and Daniel F Doak. 2002. *Quantitative conservation biology: Theory and practice of population viability analysis*. Sinauer Associates, Inc. Publishers.
- Ogden, J, GH Stewart and RB Allen. 1996. "Ecology of New Zealand *Nothofagus* forests." *The ecology and biogeography of Nothofagus forests* pp. 25–82.
- Orr-Walker, T. 2010. *Kea (Nestor notabilis) husbandry manual*. Kea Conservation Trust.
- Orr-Walker, T. and L. Roberts. 2009. "Population estimations of wild Kea (*Nestor notabilis*)."
Kea Conservation Trust .
- Orr-Walker, Tamsin, Nigel J Adams, Lorne G Roberts, Joshua R Kemp and Eric B Spurr. 2012. "Effectiveness of the bird repellents anthraquinone and d-pulegone on an endemic New Zealand parrot, the kea (*Nestor notabilis*)."
Applied Animal Behaviour Science 137(1-2):80–85.
- Otway, Nicholas M, Corey JA Bradshaw and Robert G Harcourt. 2004. "Estimating the rate of quasi-extinction of the Australian grey nurse shark (*Carcharias taurus*) population using deterministic age-and stage-classified models."
Biological Conservation 119(3):341–350.
- Pullar, Tony. 1996. *Kea (Nestor notabilis) captive management plan and husbandry manual*. Department of Conservation, Threatened Species Unit.
- R Core Team. 2019. *R: A Language and Environment for Statistical Computing*. Vienna, Austria: R Foundation for Statistical Computing.
URL: <https://www.R-project.org/>
- Robertson, C.J.R. 2007. *Atlas of bird distribution in New Zealand 1999-2004*. Ornithological Society of New Zealand.
- Simpkins, Craig E, Finnbar Lee, Breanna F Powers, Sandra Anderson, Quinn Asena, James MR Brock and George LW Perry. 2018. "Population viability analyses in New Zealand: a review."
New Zealand Journal of Ecology 42(2):107–115.
- Simpkins, Craig, George LW Perry, Andrew Glaser, Tim Allerby and Todd E Dennis. 2015. "Effects of predation by introduced mammals and mortality due to severe floods on population viability of the endangered Blue Duck (*Hymenolaimus malacorhynchos*)."
Emu-Austral Ornithology 115(2):146–157.
- Soetaert, Karline, Thomas Petzoldt and R. Setzer. 2010. "Solving Differential Equations in R: Package deSolve."
Journal of Statistical Software, Articles 33(9):1–25.
URL: <https://www.jstatsoft.org/v033/i09>
- Temple, Philip. 1996. *Book of the kea*. Hodder Moa Beckett.
- Thouless, Chris R. 1994. "Conflict between humans and elephants on private land in northern Kenya."
Oryx 28(2):119–127.
- Woodroffe, Rosie, Simon Thirgood and Alan Rabinowitz. 2005. *People and wildlife, conflict or co-existence*. Cambridge, UK: Cambridge University Press chapter The impact of human-wildlife conflict on natural systems, pp. 1–12.