

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

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This is an abstract

Keywords: These are keywords

Introduction

Humans are rapidly expanding into wildlife areas (Watson et al., 2016), causing an increase in human-induced mortality (HIM). Direct human-wildlife conflict (HWC) is widely reported because it affects humans and wildlife populations (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 (*Loxodonta africana*) or griffon vultures (*Gyps fulvus*)), and it is assumed that HIM plays a role in ~~causing and/or maintaining~~ population declines (Thouless, 1994; Landa et al., 1999; Margalida, Campión and Donázar, 2014). Unfortunately, species are often ~~at risk from~~ a myriad of threats ~~beyond HWC~~. These threats can interact making it difficult to quantify the effect of each one individually. Additionally, the expansion of a relatively minor but unmanaged mortality source (e.g. HIM), may add to a larger managed mortality source, tipping the population ~~beyond those understood by managers~~. 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 the impacts of mortality directly resultant from HWC 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 (*Nestor notabilis*) are one of New Zealand's notable 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 has led to conflict after farmers reported incidences of kea attacking sheep to eat fat from around their kidneys, occasionally leading to the 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 are still known to damage human property, and there are continued reports of kea strike on sheep, resulting in 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). ~~Though it should be noted that~~ the positive benefits to kea reproductive rates due to predator management outweighs the direct mortality caused by 1080 (Kemp et al., 2018). 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 and 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 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 to explore the qualitative differences between counterfactual scenarios, rather than making absolute quantitative predictions (Simpkins et al., 2018). ~~this has been shown to be effective in understanding the relative importance of different factors driving population dynamics (e.g. Simpkins et al., 2015).~~ A few studies have used the PVA framework to explore the relative impact of HIM on conflict-prone species (e.g. LaFever et al., 2008; Goswami, Vasudev and Oli, 2014), ~~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 was to explore how changes in **the level of** HIM, in kea habitats, will interact with introduced predator management to impact kea population dynamics, and thus determine what, if any, ~~management alterations~~ 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 predator management scenarios impact the long term viability of kea with and without additional HIM.~~

Materials and Methods

Study Species

The kea is a large, omnivorous parrot restricted to the South Island of New Zealand (Figure 1). Kea inhabit environments from ~~from~~ coastal dunes to ~~high~~ alpine peaks but are most common in high-elevation 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 is ~~that there are~~ 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 lived for 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 January, producing a single clutch with between 1 and 5 eggs. 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).

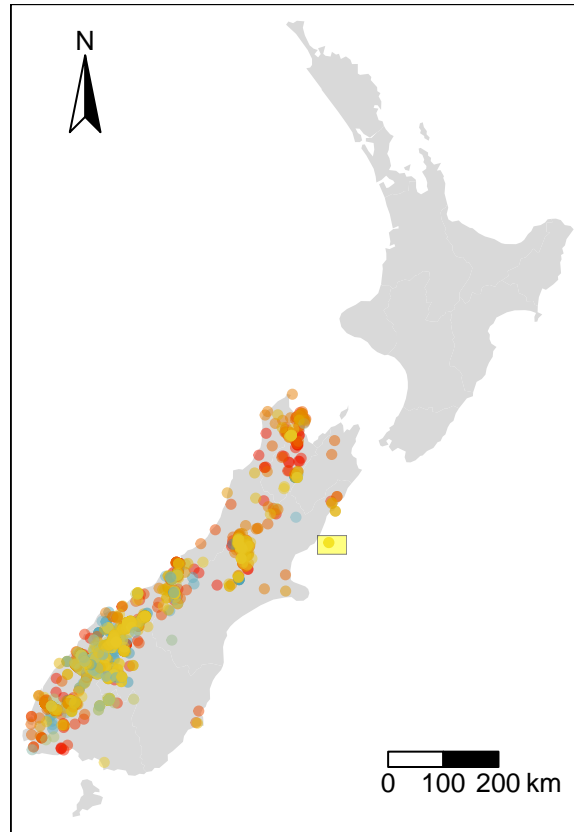


Figure 1: Kea sightings recorded since 1995 in the South Island of New Zealand (data taken from GBIF (2020)). Warmer colours indicate more recent sightings

Model Structure

We implemented a stochastic simulation model incorporating three age classes to explore the interactions between predation and 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 aggregated background, predation mortality, and HIM, with the level of predation and human-induced mortality being altered depending 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 form pair-bonds than males (Bond, Wilson and Diamond, 1991). The statistical associations between demographic parameters in the

model were perfectly correlated; although this correlation in vital rates is unlikely, it is a more conservative approach to assessing the population's viability than treating each rate as independent (Ellner et al., 2002; Ferson and Burgman, 1995). As no reliable rates of immigration and emigration were known we ~~treated~~ assumed a closed population.

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.

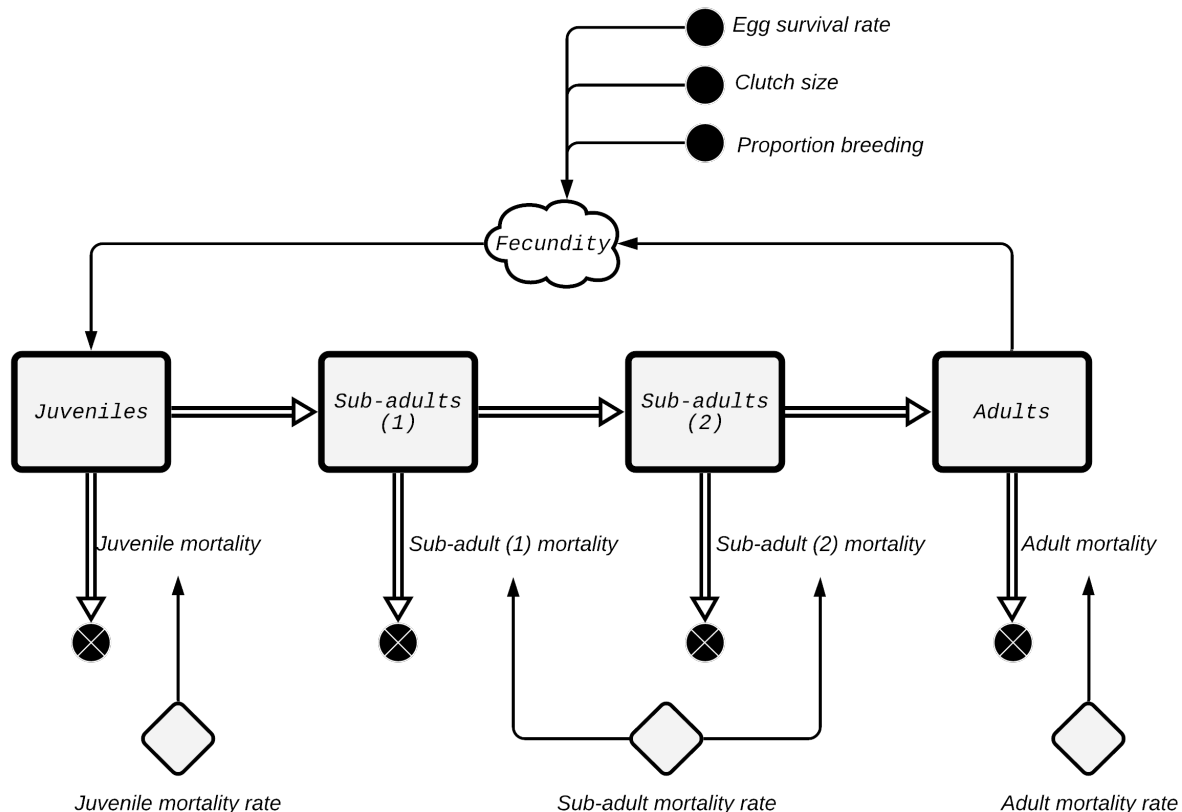


Figure 2: Overview of the kea population model. Rectangular boxes denote the population stocks for each age class, note there are two sub-adult stocks to denote that individuals remain in this class for two timesteps. The ~~large~~ double-lined arrows represent the flow of individuals. The single-lined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

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 of extinction risk. We assumed a stable age-distribution for the initial timestep.

Reproduction

Kea have a mostly monogamous breeding system (Bond, Wilson and Diamond, 1991), and 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 bounded Poisson distribution with a rate (λ) of 1.5 and with minimum and maximum values of one and eight respectively. This value was selected to generate clutch sizes similar to those observed in kea, between 1-5, taking into account that only approximately half the chicks would be female (Bond and Diamond, 1992).

Mortality

Mortality rates varied across the three age classes. Mortality rates were composed of predation, HIM, and background mortality. Predation and background mortality rates were based on the data collected by Seal et al. (1991) (Table 1.) In addition, approximately every four years a beech mast occurs (Ogden, Stewart and Allen, 1996), triggering irruptions of mammalian predators which predate kea, leading to an increase in predation mortality (Elliot, 1996; Choquenot, 2006). This masting pattern was accounted for by adding a mast event with 0.25 probability of occurring each time step. No temporal structure in the pattern of masting was assumed, giving masts a chance of occurring in successive years. During a mast event the predation rate for all age classes increased by 0.1 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 number of individuals in each age class and predation rates used in the PVA model. N is the initial number of individuals. Rates are all expressed as probability per timestep/year.

Age class	N	Current/baseline predation	Low predation	High predation
Egg	-	0.4	0.2	0.8
Juvenile	125	0.2	0.1	0.4
Sub-adult	250	0.05	0.025	0.2
Adult	125	0.1	0.05	0.3

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. We assumed HIM rates were the same for all age-classes, though no HIM was assigned to egg as kea nests are cryptic and there is minimal chance of people interacting with them. To represent different potential changes in HIM due to human population change over time we evaluated three potential population trajectories: 1) No HIM (baseline); 2) Static HIM with a constant rate of 0.015; and 3) a linear increase in HIM. The linear rate of increase was set as 0.0003 per time step. This rate was selected to match the approximate 2% national

population growth rate, assuming a one to one relationship between population size and HIM.

Linear growth was calculated as:

$$M_{t+1} = M_t + C \quad (1)$$

Where M is mortality and C is the rate of change in the population.

Model execution and analysis

Each scenario was evaluated by running 1000 simulations of 250 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 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 local univariate sensitivity analysis for clutch size and the proportion of breeding adults, which were determined to be demographic values with a high level of uncertainty. The sensitivity analysis was run using current/baseline predation rates and no HIM. Each parameter was individually varied by $\pm 10\%$ of their baseline value and the model run for 250 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

Predation management

Of the 3000 ~~runs done~~ with no additional HIM, 1029 (34.3%) resulted in extinction events (i.e. population dropping below 50 individuals). The ~~majority of these extinctions were due to the high predation scenario runs, which~~ showed a 100% extinction rate (i.e. 1000 extinction events) with a mean time until extinction of $8 \pm$ ~~a standard deviation of 0.7 time steps~~. Baseline predation

resulted in 11 extinctions, with a mean time until extinction of 106 ± 73.3 timesteps. Interestingly, the low predation scenario runs resulted in 18 extinctions, slightly higher than the baseline scenario, though this small difference is likely due to the stochastic nature of the model.

All three predation scenarios showed an initial change in population followed by a continuing period of relatively unchanging population numbers (Figure 3). The high predation scenario clearly had a higher mortality rate than replacement rate, producing a rapid loss of individuals resulting in extinction. Under baseline/current predation levels there was an initial decrease in the number of individuals, quickly reaching a stable level of approximately 350 individuals. Under low predation levels, the rate of replacement outstripped mortality initially, resulting in a period of population growth followed by a stable period with a population of approximately 500 individuals. It should be noted that even the low predation scenario did not result in a population reaching carrying capacity (1000 individuals). These time-series trends were mirrored in the mean population sizes of each predation scenario with high predation having the lowest population (6 ± 0.4), being a lot lower (Cohen D = 19.7) than the baseline/current predation scenario (335 ± 24), which was also a lot lower (Cohen D = 5.4) than the low predation scenario (527 ± 45).

The mean growth rates for the three predation scenarios showed the same trend as discussed above. The high predation scenario showed the lowest growth rate (0.072 ± 0.008), which was far lower (Cohen D = 5.5) than that of the baseline/current predation scenario (0.992 ± 0.001). There was only a notable difference (Cohen D = 0.5) between the baseline/current predation scenario growth rate, and the low predation scenario growth rate (0.984 ± 0.098), though this was likely reduced due to the long period of stable population for each scenario.

Human-induced mortality

Due to the large difference in values for the high predation scenario compared to the baseline/current and low predation scenarios it was removed from the analysis of HIM to avoid any trends being obscured.

There were only minor differences in the total number of runs going extinct between the three HIM scenarios for either baseline/current or low predation (Table 2). Notably, the no additional HIM scenarios had the highest rates of extinction (1.1% of baseline/current predation runs and 1.8% of low predation runs), likely due to the stochastic nature of the model. There was again only minor differences observed for the mean time until extinction of each run, though the rankings for the different HIM scenarios differed with predation rate.

Table 2: Percentage of total runs ending in quasi-extinction (i.e. fewer than 50 individuals) and the mean time for an extinction event to occur for each human-induced mortality scenario under baseline/current and low predation levels.

Predation level	Human-induced mortality type	Percentage of runs going extinct	Mean time-steps until extinction (\pm standard deviation)
Baseline	None	1.1	106 ± 73
Baseline	Static	0.8	146 ± 92
Baseline	Linear	0.4	219 ± 61
Low	None	1.8	119 ± 75
Low	Static	0.7	160 ± 52

Predation level	Human-induced mortality type	Percentage of runs going extinct	Mean time-steps until extinction (\pm standard deviation)
Low	Linear	1.4	128 ± 74

The general impact of the different HIM scenarios on the number of individuals in the population was the same for baseline/current and low predation levels (Figure 4). Linearly increasing HIM resulted in far lower (Cohen $D > 3.0$) population sizes for both baseline/current (211 ± 9) and low (393 ± 27) predation levels than either static or no HIM scenarios. Linearly increasing HIM also produced a notably declining population trend, resulting in larger differences with the other scenarios as model runs progressed (Figure 4). Static and no HIM scenarios produced ~~results that mirrored each other~~, though static HIM had consistently lower population sizes for both baseline/current (297 ± 20 compared to 335 ± 24 for no HIM) and low (487 ± 19 compared to 527 ± 45 for no HIM) predation levels, with these differences having a large effect size (Cohen $D = 1.730974$ and Cohen $D = 1.163097$ for baseline and low predation level scenarios respectively). The trend in mean population size is reflected in the mean growth rate produced by each scenario (Figure 6.).

Sensitivity Analysis

The model was only 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; Cohen $D = 0.69$), meaning that if the proportion of females is lower than our estimate our results may be optimistic.

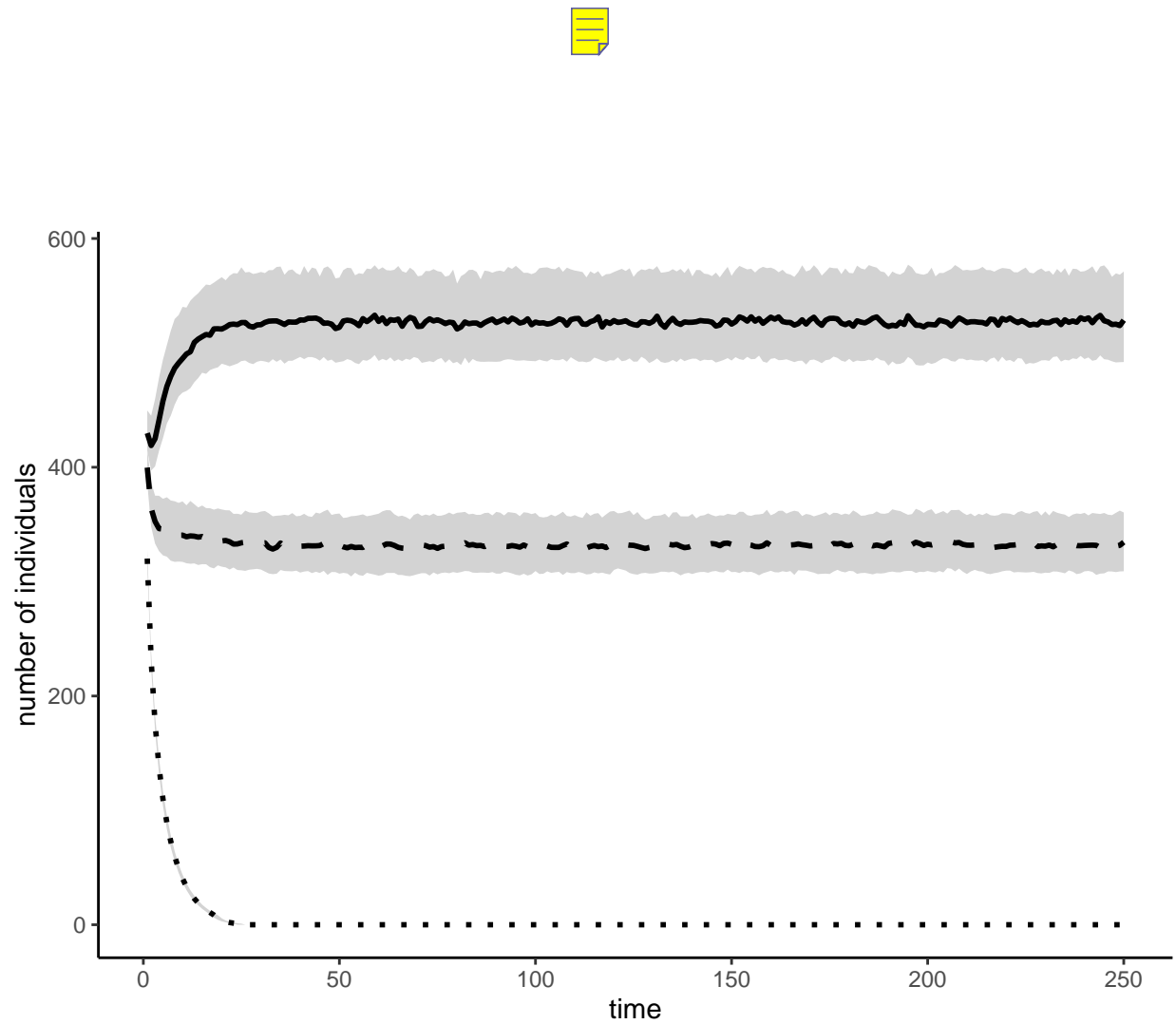


Figure 3: Change in the median number of individuals through time for each of the predation scenarios. The **solid line** is the low predation scenario; dashed line is the baseline/current predation scenario; and the dotted line is the high predation scenario. The gray bands represent the 25th to 75th quantile band

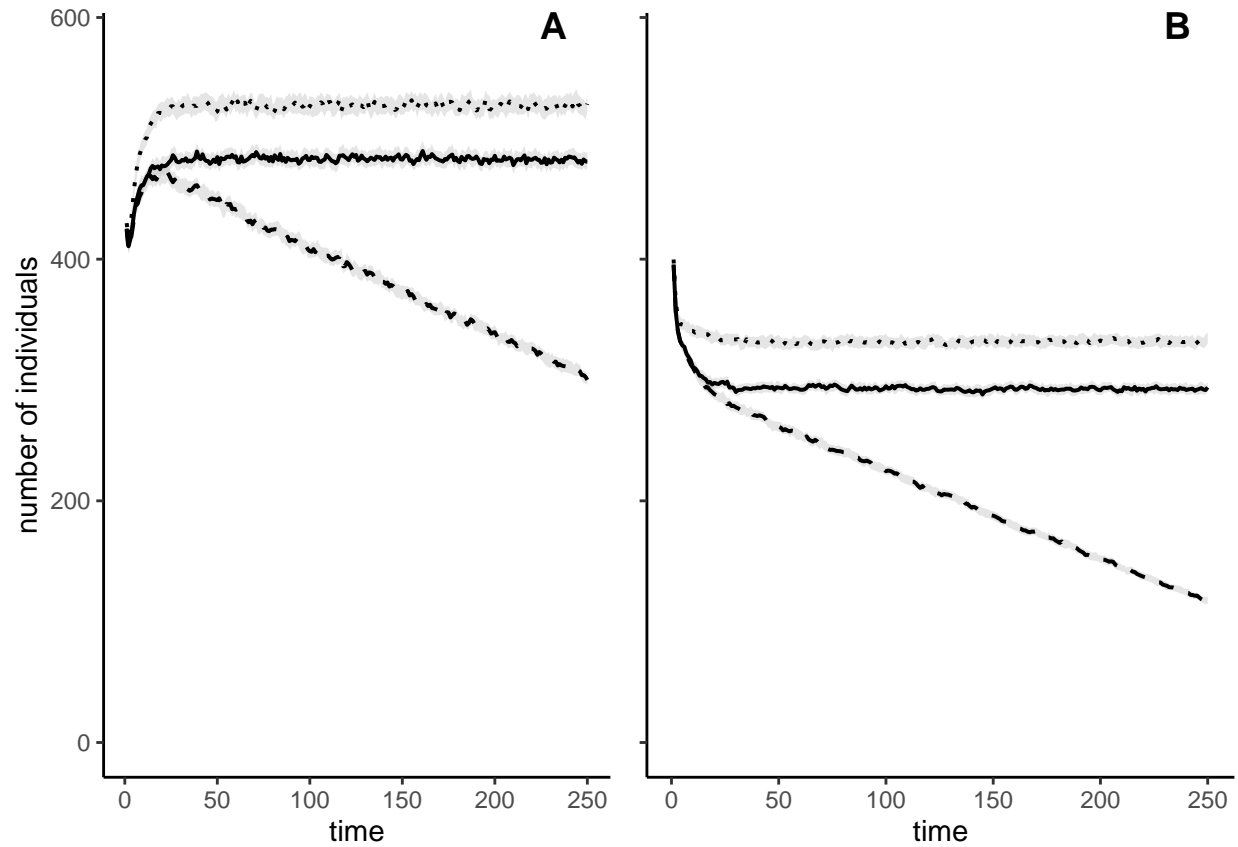


Figure 4: Change in the median number of individuals through time for each of the HIM scenarios under ~~each~~ low (A) and baseline/current (B) predation levels. The solid lines are the static HIM scenarios; dashed lines are the linearly increasing HIM scenarios; and the dotted lines are the no HIM scenarios. The gray bands represent the 25th to 75th quantile band

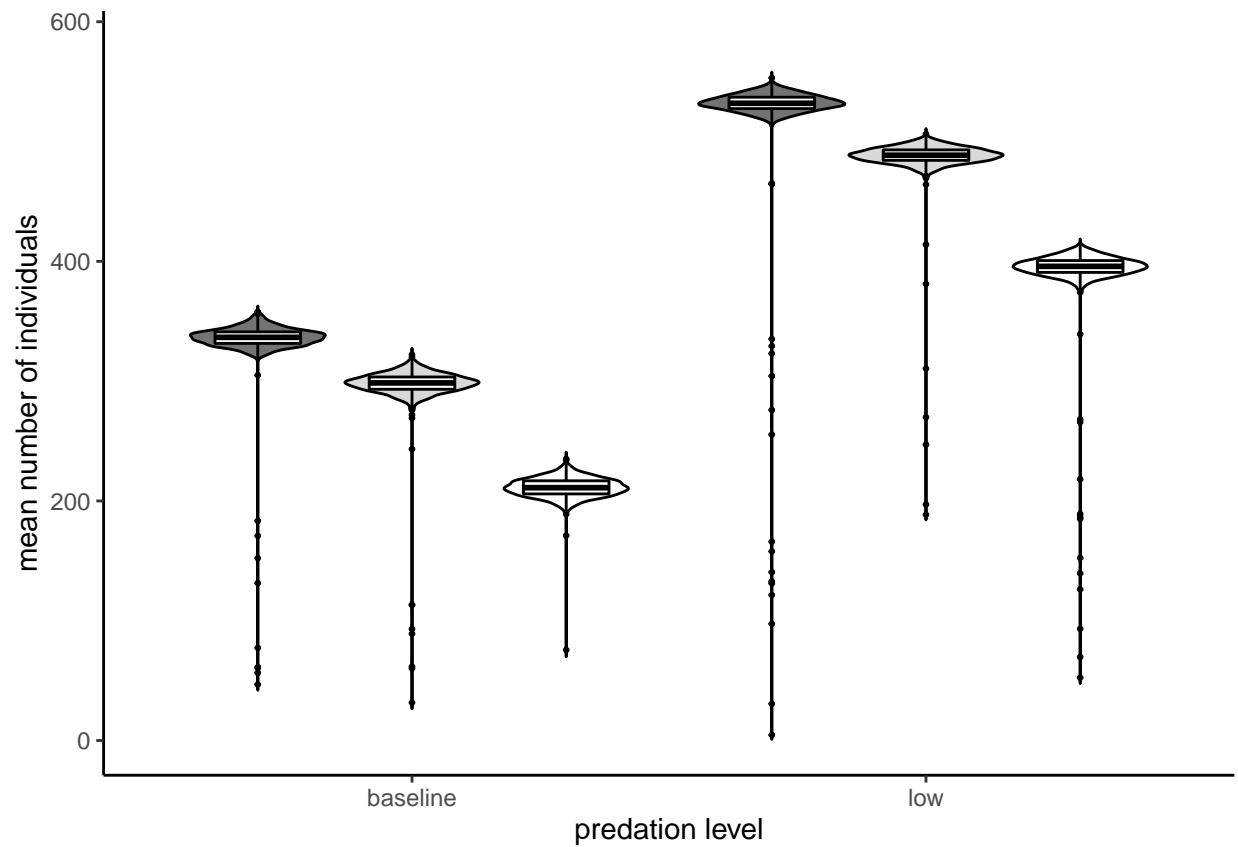


Figure 5: The mean number of individuals for each of the 1000 runs for each of the HIM scenarios under low and baseline/current predation levels. The leftmost dark gray distribution is the no HIM scenario; the center light gray distribution is the static HIM scenario; and the rightmost white distribution is the linearly increasing HIM scenario.



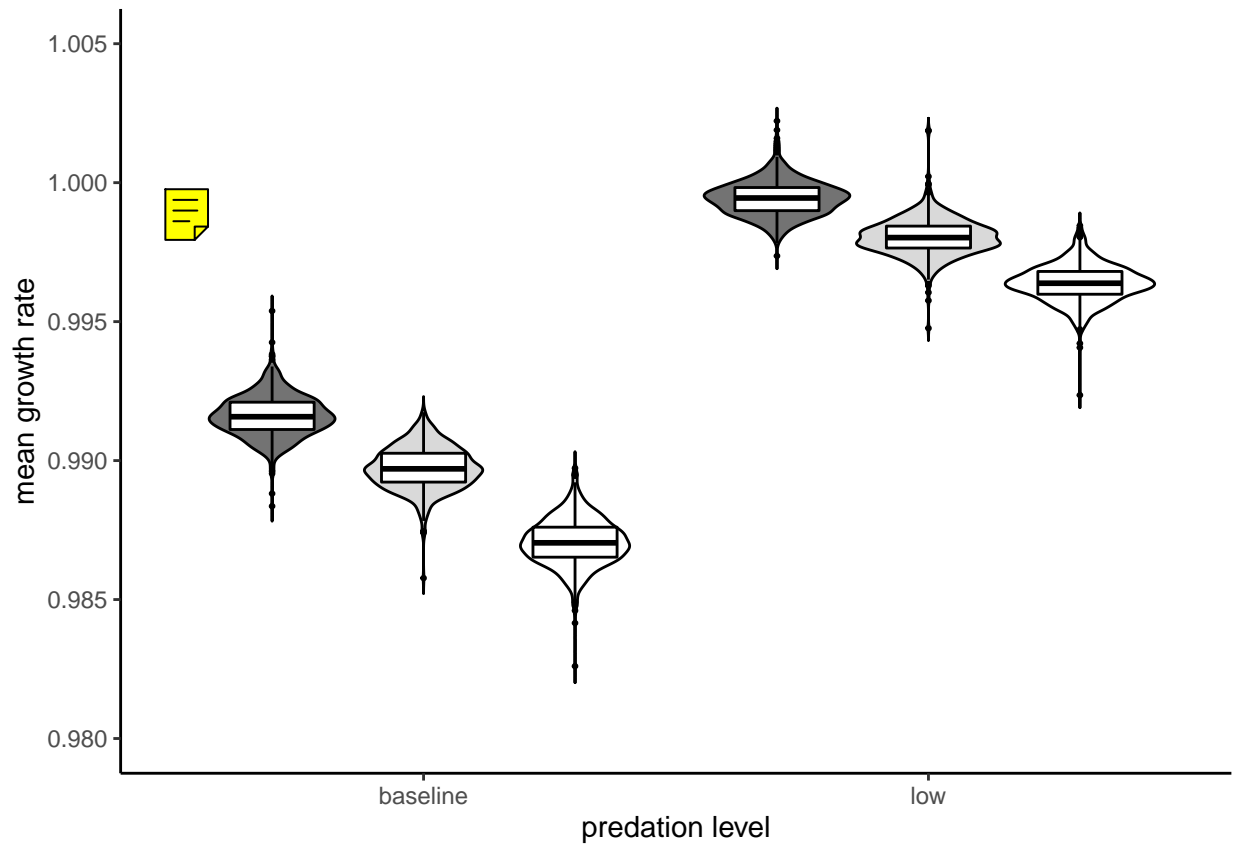


Figure 6: The mean growth rate for each of the 1000 runs for each of the HIM scenarios under low and baseline/current predation levels. The leftmost dark gray distribution is the no HIM scenario; the center light gray distribution is the static HIM scenario; and the rightmost white distribution is the linearly increasing HIM scenario.

Discussion

Our model outcomes make it clear that changes in either the predation level or the HIM rate may have dramatic effects on the population dynamics of kea. An increase in the rate of predation had the most substantial impact of the population, with all runs quickly ending in extinction (Figure 3). This result highlights the importance of ongoing predator management for the ongoing survival of kea. While a decrease in the predation level below the current managed rate, such as may be achieved under the predator free 2050 project (Russell et al., 2015), would see an increase in kea numbers, current management appears sufficient to maintain a stable population with a relatively low likelihood of extinction. Kemp et al. (2018) had similar findings when looking at kea nest survival after the application of aerial sodium fluoroacetate ("Compound 1080"), showing both a decline in nest survival in areas without pest management and a substantial increase in nest survival post predator management. Furthermore, the management of mammalian predators has been found to be the most significant factor in determining the long term survival for a number of other New Zealand bird species (e.g. mohua (Elliott, 1996), whio (Simpkins et al., 2015), and brown kiwi (Basse, McLennan and Wake, 1999)). Our study adds to the consensus that mammalian predator management is a fundamental factor in the conservation process for New Zealand.

While predation had the largest impact, it was clear that increased HIM had a substantial effect on the kea population. At the currently observed rate, HIM had only a minor negative impact on population size (and no noticeable impact on extinction risk). While any reduction in population size does decrease the resilience of the population making more susceptible to factors such as habitat loss or climate change (Keith et al., 2008). The level of change observed under current levels of HIM are likely to low to concern conservation managers. However, it is unlikely that HIM will remain at its current level, if unmanaged, given the observed growth in the human population of New Zealand, both in the number of residence and in tourism rates, and the correlation between human population and HWC events (Kretser, Sullivan and Knuth, 2008). As HIM rates increase over time they have an increasingly large negative effect on the kea population. Given the current low impact of HIM coupled with its likely growth over time, it would be best for HIM sources (e.g. vehicle collisions) to become actively managed as soon as feasible. This would allow time for management practices to improve and become embedded before HIM reaches a level of significant concern for the kea population.

The model outcomes should be viewed qualitatively, as opposed to quantitatively, as there is notable uncertainty in the values used for a number of parameters (Coulson et al., 2001). First, the base HIM are likely higher than were included in the model, because many HIM events may not be reported or detected, although this increase is difficult to quantify. Second, 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, and even potentially allow for a larger sustainable population. Third, our model outcomes were sensitive to decreases in the proportion of breeding females. As a result if the proportion of breeding females was less than our estimate or it decreased over time, the population will become more sensitive to changes in mortality, amplifying any negative impacts resulting from increases in predation rates or human populations. This uncertainty in parameter values, and thus in the precise values of the model outcomes, furthers our recommendation that management for HIM be taken while it is not a substantial threat to the population, as opposed to waiting until predictions suggest HIM will be a significant issue.

Acknowledgements

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