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Kea (*Nestor notabilis*): a review of ecology, threats, and research gaps for conservation

Kerry Weston, Josh Kemp, Kate McInnes, Joanne Aley, Tamsin Orr-Walker, Tracey Dearlove, Jamie McAulay, and Laura Young

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Cover: Kea – close up. Photographed at the Otira Gorge. *Photo: Shellie Evans* http://tikitouringnz.blogspot.co.nz/
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Kea (*Nestor notabilis*): a review of ecology, threats, and research gaps for conservation

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Abstract

Despite the iconic status of kea (*Nestor notabilis*), and their cultural significance as a taonga for Ngāi Tahu and Ngā iwi o Te Tauihu, evidence indicates a continued decline. Decades of research have contributed to a broad understanding of kea, however substantial knowledge gaps remain, especially regarding population trends and abundance, patterns of habitat use, movement, and dispersal. This likely reflects the sparse distribution of kea over a range of habitats, their mobility, social structure, and variable conspicuousness. With improved tools and technologies, some of these barriers can be overcome. Kea face a complex array of threats associated with the ongoing impacts of introduced predators, and their tendency to interact with humans and human infrastructure. Knowledge gaps remain regarding introduced predator ecology and control, lead impacts and sources, climate change impacts, and human-kea interactions. Understanding the relative impacts and interactive effects of these threats is vital to the recovery of kea, as well as measuring responses to adaptive management. It is recommended that the next step in the recovery of kea is the development of a formal recovery strategy, which includes a strategic research plan to address the knowledge gaps identified within this review.

Keywords: kea, *Nestor notabilis*, kea ecology, kea conservation, parrot conservation, endangered parrots, introduced predators New Zealand, avian lead poisoning, human wildlife conflict, aerial 1080

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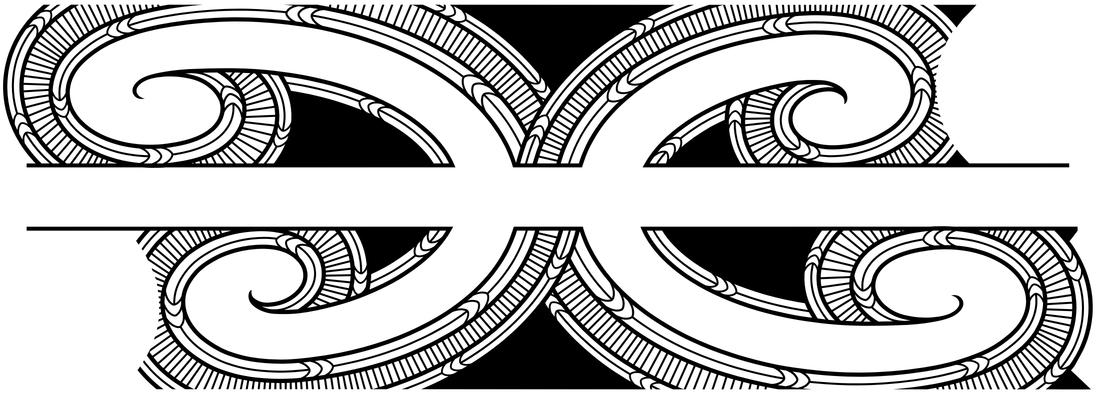

Cover illustration details

The kōwhaiwhai was designed for Te Papa Atawhai to use specifically for kea conservation such as on signage and awareness campaign messaging. The kōwhaiwhai was taken from a larger design which was based around the landscape of Te Waipounamu, such as the Southern Alps. This section was inspired from the mangōpare (hammerhead shark, *Sphyrna zygaena*) symbol which represents characteristics of the kea, including strength, strong will, and a fighting spirit. This is bordered by an intricate design representing raumoa, symbolising the feathers of manu (bird).

About the artist

The kōwhaiwhai was designed by Fayne Robinson (Ngāi Tahu, Ngāti Māmoe, Ngāti Apa ki te Rā Tō), a carver based in Ōtautahi Christchurch. Fayne trained at the New Zealand Māori Arts and Crafts Institute, Rotorua. He has worked on wharenui as a master carver, has exhibited both nationally and internationally, and has many public installations. His works are also held in the collections of the Museum of New Zealand Te Papa Tongarewa.

1. Introduction

Kea (*Nestor notabilis*) are an endangered taonga, of special cultural significance to Ngāi Tahu and Ngā iwi o Te Tauihu (Rangitāne o Wairau, Ngāti Kuia, Ngāti Apa ki te Rā Tō, Ngāti Toa, Ngāti Rārua, Ngāti Koata, Ngāti Tama, Te Ātiawa o Te Waka-a-Māui). Despite this significance, kea have a long history of human persecution, and remain sparsely distributed throughout Te Waipounamu.

Kea face an array of threats associated with the ongoing impacts of introduced predators and risks resulting from their propensity to interact with human resources and infrastructure.

This document brings together a summary of current knowledge of kea and the threats to population recovery, and identifies key gaps in this knowledge. The primary aim is to identify knowledge gaps that need to be addressed before an effective kea recovery strategy can be fully implemented.

2. Understanding kea

2.1 Abundance and population trends

2.1.1 Historic population size

Kea are thought to have been more abundant historically than they are today. Genetic work suggests an early Holocene (11.6 kya) population contraction in response to climate warming and changes in habitat distribution (Dussex et al. 2015; Dussex et al. 2014). Evidence for at least one pre-human population contraction is further supported by the discovery of Pleistocene kea fossils in Te Ika-a-Māui (Holdaway and Worthy 1993) and areas of Te Waipounamu where kea are now rare or absent such as central, eastern, and coastal Otago, central Westland (Punakaiki area) and eastern Southland (Worthy 1998; Worthy and Grant-Mackie 2003; Worthy and Holdaway 1993). The kea population underwent a major human-induced decline between the late 1800s and 1970s when an estimated 117,000–150,000 individuals were culled under a government bounty scheme prompted by attacks on sheep (Reid 2019; Temple 1996). Large numbers of individuals were removed from some regions, with > 6,000 kea bounties claimed in the region around Queenstown (Lake County, 10,000 km^2) between 1943–1945 alone (Cunningham 1947).

2.1.2 Threat status and legal protection

Since 1986, kea have been a fully protected species under the Wildlife Act 1953. Kea have special cultural significance and importance to Ngāi Tahu and Ngā iwi o Te Tauihu, as reflected within the Ngāi Tahu Claims Settlement Act 1998. This act formally recognised kea as a taonga species to Ngāi Tahu. In 2013, the New Zealand threat status of kea was changed from At Risk (Naturally Uncommon) to Threatened (Nationally Endangered) based on an estimated population size of 1,000–5,000 mature individuals and a predicted population decline of 50–70% over three generations (Robertson et al. 2013). The IUCN threat status of kea also reflects a continued declining population trajectory, with a transition from Least Concern in 1988, to Near Threatened in 1994, Vulnerable in 2000, and Endangered in 2017 (IUCN 2023).

2.1.3 Past and present distribution

The kea population is sparsely distributed across its current range, of approximately 3.5 million hectares, from Farewell Spit at the north of Te Waipounamu through to southern Fiordland (Robertson et al. 2007). Isolated, unconfirmed records of birds, post-European settlement, in Te Ika-a-Māui have been recorded from 1942 and 1974 – though it is likely these records refer to escapes from captivity or misidentification of the closely related kākā (Cunningham 1974).

Kea are most widespread along the western and central slopes of the Southern Alps/Kā Tiritiri o Te Moana, but are also found northwards into Kahurangi National Park and Abel Tasman, and south to the southern reaches of Fiordland National Park and a few near-shore islands (https://www.keadatabase.nz). Kea are less abundant in eastern areas of Marlborough, Canterbury and Southland, with isolated sightings within the Seaward Kaikōura Ranges and Richmond Range. Kea are absent from the Marlborough Sounds, Catlins, Blue Mountains, and Stewart Island/Rakiura (Robertson et al. 2007).

Although often referred to as a 'mountain species', kea occupy a broad altitudinal distribution ranging from sea level up to approximately 2100 m asl along the slopes of the Southern Alps and associated ranges (Robertson et al. 2007). Fossil evidence from the Holocene in lowland Canterbury (Holdaway and Worthy 1997) and from several sites across Te Ika-a-Māui suggest that the present distribution of kea is relictual (Holdaway and Worthy 1993; Tennyson et al. 2014). Recent studies consider the present distribution of kea may instead reflect where kea have survived human-caused extinctions or represent a suite of more generalised habitat types that kea have exploited to escape competition with more specialist species (Dussex et al. 2015; Martini et al. 2021).

Limited knowledge of early Māori interactions with kea has been perceived to indicate that they were rare during early human settlement across Te Waipounamu, existing in only small, isolated populations (Bond and Diamond 2019). Kea were considered kaitiaki (guardians) of the high country for the Waitaha Māori during their search for pounamu (greenstone) (Orbell 2003). Ngāi Tahu travellers also crossed most of the major passes of the Southern Alps in search of pounamu, so it is likely that they had contact with kea, but beyond the naming of the bird for the long high-pitched cry, kea feature rarely in written Māori mythology and traditions. While bird remains are common in Māori middens, the only midden site thought to contain kea remains is from Lee Island in Fiordland and dates back to the sixteenth to eighteenth centuries (although it also contained a small amount of twentieth-century material) (Diamond and Bond 1999). This lack of presence in middens may be because kea were described as too 'tough and lean' for eating, as well as having a strange smell when cooked (Riley 2001). The feathers, however, continue to be prized for their use in kākahu (cloaks). Ngāi Tahu kaumātua have recorded several regions in southern Te Waipounamu (e.g. Kawarau, Makarora) as a kāinga mahinga kai (harvesting places) where kea were gathered (Cain 2020). Kea were once caught in the alpine regions of Te Ika-a-Māui, however, which indicates at least a period of co-inhabitation with Māori prior to their subsequent extinction in the north (Riley 2001). Introduced mammals kiore (*Rattus exulans*) and kurī (*Canis familiaris*) arrived in New Zealand alongside the first humans and the impacts of these species on kea is unknown.

For a few decades following the first encounters of European settlers with kea in 1856 (Gould 1856) kea were considered relatively rare by naturalists and described as 'essentially a mountain species', recorded only from mountainous areas south of Arthur's Pass with strongholds particularly noted in parts of Otago (Diamond and Bond 1999). Following intensive settlement of Te Waipounamu high country by early runholders, kea appeared to rapidly expand in numbers (Diamond and Bond 1999). It was suggested that they expanded their range into the northern parts of Te Waipounamu post European settlement (Marriner 1909; Myers 1924; Oliver 1955). However, it is generally agreed more likely that kea already inhabited northern areas, and increased number of records probably simply reflected increased number of observers, combined with the effect of bounties stimulating reports (Brejaart 1988; Holdaway and Worthy 1993).

2.1.4 Estimating population size and trends

Methods for measuring the population size and trends of kea are not well developed. A sparse distribution over a range of habitats, high mobility, complex social structure, and variable conspicuousness makes kea very difficult to count and therefore monitor. Methods commonly employed for population monitoring such as territory mapping, capture-mark-recapture, call counts, and distance sampling are not well suited to kea. Attempts to apply these methods to kea becomes prohibitively expensive and/or violates the statistical assumptions of the methods.

Total population estimates for kea numbers are wide ranging, from the commonly cited but unsubstantiated 1,000–5,000 individuals (Anderson 1986; Birdlife International 2023) to as high as 15,000 individuals (Bond and Diamond 1992). These studies all extrapolate results of localised studies to the wider species' range, usually from/near sites of human activity. The former estimates appear to be based on one adult female per 2,000 hectares of forest, giving a total of 4,000 mature individuals. Productivity estimates then predict one juvenile for every breeding pair, giving a total of about 6,000 birds (Birdlife International 2023; Josh Kemp, pers. obs.). The census size (N_c) of kea inferred from contemporary effective size (N_e) estimated using genetic approaches is also within the range of the current lower estimates for kea abundance (1,300–6,200) (Dussex and Robertson 2018). Early attempts at modelling kea population growth and viability recognised a lack of essential data on kea demography, including annual survival and productivity rates and total population size (Seal et al. 1993).

Capture-mark-recapture

Initial capture-mark-recapture studies were undertaken at places where kea congregate to exploit human foods, such as landfills and ski fields (Bond and Diamond 1992; Clarke 1970; Elliott and Kemp 2004; Jackson 1960). These studies lacked statistical power due to small sample sizes, sex and age biases in scrounging propensity, and the difficulty in defining the sampling area (for example, accounting for the distances kea will travel to scrounge). Therefore, the value of the density estimates obtained is questionable and it is perhaps not surprising that no indication of population change was found. For example, Bond and Diamond (1992) estimated a population density of between 0.018 and 0.040 birds per hectare in the vicinity of a refuse dump within Arthur's Pass National Park, with the number of adults appearing quite stable across three years. Thirty years earlier, Jackson (1960) estimated the density of kea at the nearby Temple Basin (another area where kea commonly scrounge) at about 0.032 birds per hectare. In contrast, Clarke (1970) estimated a resident density of about 0.004 birds per hectare at the remote Cupola Creek within Nelson Lakes National Park between 1964 and 1966. All of these estimates have problems with violation of assumptions and applicability to the wider population.

Territory mapping

Territory mapping/census of adult female kea within Nelson Lakes National Park in 1999 and 2011 indicated a 70% decline in abundance over 12 years (Department of Conservation [DOC] & Kea Conservation Trust [KCT], unpubl. data). This decline may have been a consequence of extended, high magnitude predator irruptions recorded at Nelson Lakes and within most other eastern beech forests during 2000–2002, which was in the interval between kea censuses (Kemp et al. 2022). The density of adult female kea in these two censuses was one per 550 ha and one per 2,750 ha, respectively. These territory mapping exercises were labour intensive and aided by a low density of kea and relatively navigable terrain. Attempts at territory mapping elsewhere (Hawdon Valley and Poulter Valley, Arthur's Pass, and Borland Range, Fiordland) were not completed, largely due to complications arising from more difficult terrain and higher kea densities.

Sex-ratio

Male kea are over-represented among capture and observation data (Bond and Diamond 1992; DOC, unpubl. data). Whether this reflects a true sex-bias within the kea population, or is a behavioural effect is unclear, though the latter seems most likely. It has been hypothesised that adult male kea are bolder and more exploratory than females because they are required to provide food for the female kea and nestlings during breeding (Diamond and Bond 1999). Female kākā (*N. meridionalis*) are known to be less susceptible to capture than males, however a significantly skewed sex ratio toward male kākā in sites without effective predator control also reflects the fact that female kākā suffer higher mortality due to predation at the nest than males (Greene et al. 2004; Greene and Fraser 1998). Unlike kākā, kea nest monitoring data does not indicate adult female mortality at the nest. Of 203 kea broods monitored between 1996 and 2019, 93 failed and in none of these cases was the female found dead in the nest; instead, females were consistently confirmed alive after nest failure by visual sighting of coloured leg bands (DOC, unpubl. data). Avoidance of predation on the nest may be enabled by the physical characteristics of kea nest cavities, which often have more than one entrance/exit through which the adult female can escape. Also, an adult female kea, at about 800 g, presents a relatively challenging prey item compared to the eggs and nestlings. There is no other obvious mechanism for a demographic sex-bias in kea. The sex ratio at hatching is 1:1 (Josh Kemp, pers. obs.) and radio-tracking studies show no difference in survivorship of male and female kea (Kemp et al. 2022). Given that the evidence to date indicates the observed skewed-sex ratio is likely to be a behavioural artefact, sex-ratio is unlikely to be an accurate indicator of kea population health.

Five-minute bird counts

Existing long-term call-count datasets indicate this method has inadequate power to detect significant population trends in kea when sampling is undertaken on a spatial and temporal scale suitable for other forest birds (O'Donnell and Hoare 2012). For example, annual 5-minute call counts have been carried out since 1998 across c. 900 ha in the mid Landsborough Valley, South Westland. The sampling regime comprises 1–2 counts per year at each of 113 forest counting stations, as part of a project measuring long-term responses of bird populations to sustained predator control (O'Donnell 2019). These counts detected no significant change in call rates within the valley over 21 years from 1998–2019 (Colin O'Donnell, DOC, unpubl. data). Given the low detection rates of kea during the call-counts, a retrospective power analysis gives the survey regime less than 10% power to detect a 5% per annum change in call rate. That is, given the existing sampling effort, there is a 90% chance that a 5% per annum change would be obscured by sampling 'noise' and it would take approximately 75 years to detect change if it were occurring.

Acoustic recorders

Kea call counts using remote audio recorders show a similar lack of power to detect population trends in kea. A total of 300 audio recorders were located across South Westland and activated five times per year for 5-min counts between 2010–2016, to measure responses in forest birds to sustained predator control. Again, no significant change in kea call rates was detected across the survey area over the seven-year study period (Graeme Elliott, DOC, pers. comm.). Statistical power could be increased to give 80% probability of detecting a 5% change in call rate per year if the sampling period was increased to 15 years and the sampling effort was doubled, however this would be very expensive and provide only a localised measure (Graeme Elliott, DOC, unpubl. data). It is possible that changes in site occupancy (recorders that detect presence of kea calls daily) would better reflect changes in density and that further analysis of existing datasets could yet yield useful trends. Machine learning approaches could also help separate out sex groupings or individual kea within bulk recordings to add granularity to these existing data.

Birds New Zealand Atlas

National atlases of bird distribution compiled by Birds New Zealand (OSNZ) comprise the only multi-decade dataset of the spatial distribution of kea across their range (Bull et al. 1985; Robertson et al. 2007). Field surveys for the first atlas were conducted 1969–1979 and for the second atlas 1999–2004. Walker and Monks (2018) used occupancy modelling to generate a probability of local occupancy for kea within 10 km x 10 km grid squares across Te Waipounamu, enabling a comparison of median estimates between the two survey periods. These estimates indicated a decline in occupancy across the 20-years separating the two survey periods. Although occupancy estimates based on distribution data are useful for assessment of large-scale changes, this approach has limited utility for measuring population trends of kea in response to management.

Kea Survey Tool

The most recent attempt at measuring population trends of kea across their entire range is a web-based citizen-science approach, the Kea Survey Tool, which was established in 2019 (https://survey.keadatabase.nz/). As with the bird atlas, an advantage of the Kea Survey Tool is that it covers a large spatial scale – the entirety of Te Waipounamu. The kea surveys are carried out by backcountry users whilst tramping, camping, hunting, etc., and provides two metrics: 1) the probability that kea are encountered, per hour, and 2) the maximum group/flock size seen per day. Survey hours and kea encounters are tagged to a location, using a grid of 5 km x 5 km squares. The grid is based on the New Zealand Bird Atlas grid, with the intention of importing OSNZ surveys into the Kea Survey Tool. Power analysis based on the first three years pilot study indicates that detecting a 5% per annum change in kea encounter probability in South Westland over a ten-year timeframe, for example, will require about 75 surveys per year. Timeframes could be shortened and/or sensitivity increased by increasing the number of surveys each year (Josh Kemp, pers. obs.).

Preliminary analyses indicate strong spatial patterns, with higher kea encounter rates and larger flock sizes in the southwest of Te Waipounamu and fewer in the east of the species' range (Josh Kemp, pers. obs.). However, variation in backcountry visitation rates means that survey effort is under-represented in some areas, for example the Kaikōura Ranges. Ways of addressing this sampling bias need to be considered and further research exploring the utility of this tool is warranted.

As detectability of kea can fluctuate seasonally and spatially in ways that are not yet well understood, there is also a need to investigate alternative methods and analytical techniques that are commonly used on highly mobile fauna elsewhere – in particular those which can account more formally for all components of incomplete detection, including the probabilities of presence and availability (i.e. Berigan et al. 2019; Emmet et al. 2021; Nichols et al. 2008; MacKenzie et al. 2002; MacKenzie et al. 2005; MacKenzie and Nichols 2004; Thompson 2004).

2.2 Genetic structuring

Genetic work on kea indicates only moderate population structuring across their range, but nonetheless identifies 2–3 distinct, geographically separated genetic clusters (Dussex et al. 2014; Stubbs 2022). Dussex et al. (2014) identified three genetic clusters, representing the northern, central, and southern regions of Te Waipounamu. A northern cluster includes kea from Golden Bay/Mohua, Kahurangi, Nelson Lakes, Kaikōura, and Arthur's Pass. A central cluster comprises kea from Westland and Aoraki, and a southern cluster includes kea from Aspiring and Fiordland National Parks. Dussex et al. (2014) proposed these genetic clusters are likely the result of recolonisation processes following the last glacial maximum (about 20,000 years ago) and not human-induced fragmentation. Stubbs (2022) used a higher resolution genome-wide dataset and identified two genetic clusters, northern (primarily north of Aoraki/Mt Cook) and southern (primarily south of Milford Sound/Piopiotahi) with a clinal pattern of genetic variation from north to south through Te Waipounamu.

Given the limited extent and recent evolutionary origin of genetic structuring in kea, Dussex et al. (2014) suggest that each genetic cluster need not be considered as an independent conservation unit. Based on large-scale gene flow estimates, the Te Waipounamu kea population comprises a single entity, comprised of geographically separated 'sub-populations' connected by dispersal.

2.3 Kea ecology

2.3.1 Breeding ecology

Kea nest on the ground, almost always below the treeline and predominantly in upland beech or lowland podocarp forest. Nests are located under a boulder, in a crevice, or among the roots of a tree (Elliott and Kemp 2004; Jackson 1963). Active kea nest sites are sparsely distributed within the landscape, with three studies reporting a density of around one nest per four square kilometres, though the representativeness of these studies is unknown (Jackson 1960; Bond and Diamond 1992; Elliott and Kemp 1999). Of 196 active nest cavities located since 1992, nest altitude ranged from just above sea level in South Westland (30 m a.s.l) through to 1350 m a.s.l in Nelson Lakes National Park (DOC, unpubl. data).

Pulses of breeding of congeneric kākā (*N. meridionalis*) and confamiliar kākāpō (*Strigops habroptilus*) coincide with increased food supplies associated with beech flowering and rimu fruiting respectively (Fidler et al. 2008; Powlesland et al. 2009). It appears that kea breeding is not contingent on similar resource cues, as attempted nesting by at least some kea has been observed in most years. Occurrences of the same pairs breeding in several successive years have been reported, and pair bonds lasting up to a decade, possibly more, have been observed (Elliott and Kemp 2004; Jackson 1963; Kemp et al. 2018). Occasional breaking of pair bonds has also been observed (Josh Kemp, pers. obs.). Breeding pairs usually attempt only one nest per year, though sometimes attempt a second clutch if the first one failed early in the season. On rare occasions (2 of 164 monitored nest attempts), a second clutch is produced following the successful fledging of the first (DOC, unpubl. data).

The brooding period spans about four months (c. 116 days) from egg-laying to fledging. The majority of egg laying occurs in July-August though has been recorded through until January, particularly in the case of renesting (Elliott and Kemp 2004; Jackson 1963). Data collected to date indicates that there may be geographical variation in the timing of nest initiation, with egg-laying commencing earlier in South Westland relative to other areas such as Kahurangi, Nelson Lakes, or Arthur's Pass (DOC, unpubl. data). Clutch size varies between 1–5 eggs, with 3 eggs laid on average (n = 31) (DOC and KCT, unpubl. data). The female kea carries out all of the incubation and the male provides her with food during this period. Once fledged, the young are dependent on the adult male for a further 2–6 weeks (Diamond and Bond 1991; Jackson 1963).

Of the 164 active nesting events monitored between 1993–2015, throughout the species' range, 51% (83/164) were successful (i.e. fledged one or more young). The number of fledglings recorded from successful nests varied from 1–4, with 1.9 fledglings on average (DOC, unpubl.

data). Proportionally, 28% fledged one young, 59% fledged two, 12% fledged three, and only one nest was observed that fledged four young (DOC, unpubl. data).

Similar numbers of fledglings have been reported from successful nests within upland beech forest (mean = 1.6, *n* = 20) (Jackson 1963) and lowland podocarp (mean = 1.8, *n* = 25) (Kemp et al. 2018). Further studies of nesting effort and nesting success are required in different habitats to further understand spatial and annual variation in productivity and nest survival.

2.3.2 Habitat and diet

In wetter regions west of the Southern Alps, kea occupy lowland rimu and mixed rimu-beech forest, through to upland rātā-kāmahi (*Metrosideros* spp, *Weinmannia racemosa*) forest, upland *Olearia* scrub and upland silver beech forest (Kemp et al. 2018; O'Donnell and Dilks 1986). Where kea persist in eastern regions, they are largely confined to montane pure-beech-forested valleys (Kemp et al. 2022). Adjacent subalpine and alpine grassland, scree and herb field ecosystems are also used on both sides of the alps, predominantly for foraging and socialising (Diamond and Bond 1999; Young et al. 2012). Production forests (predominantly *Pinus radiata*) and farms of various types also adjoin kea habitat throughout their range. Kea sightings are reported from these areas, but as a proportion of their current range, these comprise minor habitats (https://www.keadatabase.nz).

Kea have an omnivorous, generalist diet and have been recorded feeding on over a hundred different species of plants and animals across a range of habitats, from sea level to above the treeline (Aitken et al. 2022; Brejaart 1988; Clarke 1970; O'Donnell and Dilks 1994; Schwing 2010; Young et al. 2012). Kea obtain a substantial proportion of their diet underground and have been observed digging in soil and clay for several hours to access the underground tubers of New Zealand native orchids (*Gastrodea* spp.) (Josh Kemp, pers. obs.). Excavations of 10–15 cm depth, attributable to kea digging, are commonly found within Te Waipounamu beech forests, sometimes at high density and over large areas (e.g. >100 square metres) (Graeme Elliott pers. comm). The majority of feeding activity occurs during the early morning or late afternoon and early evening (Brejaart 1988).

Stable isotope and faecal analyses indicate that kea consume plants as a primary food source throughout their range, with the exception of within the lowland rainforest of Westland where invertebrates comprise a larger proportion of their diet (Greer et al. 2015; McLean 2023). Kea from western rainforest also have a wider foraging niche than those living in dry eastern regions, which may reflect a difference in food availability (McLean 2023). Differential foraging strategies have also been linked to morphological differences among sub-populations, with lowland rainforest birds having longer bills and larger heads than those breeding in high-altitude montane regions (Greer 2015). Isotope analysis of kea feathers held in museums collected from the 1880s to 2000s suggests that historically, the diet of kea may have been less variable across the population (Wehi et al. 2022).

Among the sexes, the upper bill of kea is 12–14% longer in males, with increasing evidence that this is an adaption for sexual specialisation in foraging (Bond et al. 1991; Greer 2015). There is a tendency for male kea to forage at higher trophic levels (Greer 2015; Wehi et al. 2022; McLean 2023), which may reflect more dominant foraging or scavenging behaviours among males (Reid 2019; Wehi et al. 2022). Differences in foraging behaviours also appear to become more pronounced with age, likely due to the adult male's role in provisioning females and offspring during nesting (McLean 2023).

In spring and summer, flocks of kea are often observed feeding within alpine grasslands and herbfields, consuming a larger proportion of invertebrates during spring and appearing to shift to mainly fruits during summer and autumn when these become available at higher altitudes (Greer et al. 2015; Young et al. 2012). In contrast to parrots in general, which usually destroy the seeds they consume, kea tend to ingest whole seeds and defecate them intact.

Consequently, kea play an important role in seed dispersal for alpine plants (Young et al. 2012). A shift to a more foliage-based diet occurs during winter when kea generally retreat to lower altitudes (Greer et al. 2015).

Kea also exploit seasonally available nectar sources, such as harakeke/flax (*Phormium tenax)* or southern rātā (*Metrosideros umbellata*) (O'Donnell and Dilks 1994). Flocks of kea have been observed feeding on harakeke nectar when flowering prolifically including along coastal margins within South Westland and Golden Bay (https://www.keadatabase.nz).

2.3.3 Home range and movement ecology

Movements of kea are still poorly understood, particularly those of juvenile birds up until the age of sexual maturity at >3 years of age for females and 4–5 years of age for males (Diamond and Bond 1999). Juvenile kea can be highly mobile, sometimes travelling long distances. At Arthur's Pass in the 1960s, commutes of 5 km in one day to scrounge food at an open landfill were frequently observed, with one movement of 45 km also recorded (Jackson 1960). Data obtained from GPS loggers on 10 male kea near Arthur's Pass village between 2012–2014 showed a degree of association between kea movements and human settlements and infrastructure (Kennedy et al. 2015). All kea preferentially selected human areas within their home ranges, and habitat use of human areas was disproportionately higher than expected on the basis of availability. Sexually immature birds spent considerably more time closer to human areas than did sexually mature males who were likely provisioning mates (Kennedy et al. 2015). However, the kea in this study were all caught at the Arthur's Pass scrounge sites, so it's not surprising that they continued to visit them.

In recent years, kea flocks comprising up to approximately 20–30 individuals have been recorded during winter at alpine hut sites in South Westland (e.g., Chancellor and Castle Rocks Huts). The origin and demographics of these flocks is unknown, though they appeared to be comprised of predominantly juvenile/pre-breeding age birds accompanied by one or two adults (Rose Lanman, DOC, pers. comm.). Weekly turnover of individual birds coming through these sites also appeared to be high, with colour banding data showing an ongoing high ratio of unbanded to banded birds despite frequent banding trips (DOC, unpubl. data). Flocks of kea are observed in all seasons, though little is currently known about the movements and demographics of these flocks and how they change.

Early studies indicated kea became more sedentary on breeding territories once mated, with movements centred around their nests and nearby roosts. Breeding males at Arthur's Pass were thought to typically travel within c. 1–3 km of the nest to forage and provision nesting females whilst nesting females were thought to remain close to the nest year-round (Jackson 1960). Elliott and Kemp (2004) observed nesting male kea regularly visiting Rainbow Ski Area on the St. Arnaud Range, with nests up to 6 km away, though adult females only rarely visited the ski area and those that did nested within 3 km of it. At Aoraki/Mount Cook, breeding adults rarely ventured more than 1.5 km from their nest site, while non-breeding animals ranged over about 6 km of the valley systems (Higgins 1999). A pair of kea will join a flock while it is in their territory or that of neighbouring pairs, but the pair returns to its own roost each night (Jackson 1960). However, more recent radio-tracking and GPS movement data suggest that adults may move considerably further distances than originally thought (DOC, unpubl. data). A full understanding of what these movements mean in terms of kea home range or territories and the extent of movement both within and outside of the breeding season is yet to be developed. For example, it is unknown to what extent kea breeding in lowland forest also utilise montane and alpine areas for foraging throughout the year.

Altitudinal movements probably occur in response to weather, snow conditions, and availability of food sources; social, or other reasons (Brejaart 1988; Diamond and Bond 1999). The patterns and frequency of these movements are not known, but there are several examples of pre-breeding kea banded within lowland forest at Ōkārito in South Westland, reappearing months later on remote cameras in alpine habitat 30–40 km away (DOC and ZIP, unpubl. data).

2.3.4 Pre-breeding dispersal

After fledging the nest, young birds are dependent on the male kea for food for a further 2–6 weeks and have been observed to remain with their parents for up to 8 weeks, and possibly longer (Diamond and Bond 1999; Jackson 1963). By their second summer, many juveniles have dispersed from their natal ranges and appear to move around in loosely organised flocks (Diamond and Bond 1999). Banding and radio-tracking data have revealed individual juveniles often move very long distances. For example, a male fledgling banded at Deaths Corner (the Otira Viaduct Lookout,) near Arthur's Pass, was recorded c. 100 km away just a few months later where he was observed by a hunter in South Westland with a group of about 15 other mainly young birds (https://keadatabase.nz/birds/romano). In 2022, a pre-breeding male banded as a fledgling in 2017 near Arthur's Pass was re-sighted c. 500 km away near Farewell Spit (https://keadatabase.nz/birds/wananga). This sighting was preceded by several days of sightings along the West Coast and inland towards Arthur's Pass.

Kea have also occasionally been recorded from the Port Hills in Christchurch (Brejaart 1988) – most recently in 2023 when two single birds were sighted at several locations across the city (https://keadatabase.nz).

Banding data has also revealed dispersal movements across the main divide. For example, a fledgling banded at Fox Glacier/Te Moeka o Tuawe in South Westland was resighted seven months later at Aoraki/Mount Cook (Higgins 1999). Recently, a juvenile female banded in the Upper Rakaia in 2021 was resighted four months later west of the main divide near the Adams Range (https://keadatabase.nz/birds/couloir).

2.4 Understanding kea – knowledge gaps

Based on the review above, the following are gaps in our knowledge that require further research and tool development to assist in recovery actions:

- Methods for measuring population size and trends, and associated data platforms to collate and manage data – (e.g., kea survey tool and sightings database development, call counts, transects, acoustic monitoring, RFID monitoring and occupancy modelling).
- Population models based on up-to-date demographic data from across kea range i.e. sex ratios, age at first breeding, annual productivity rates, proportion of population breeding, annual age-specific survival rates, and dispersal behaviour. A multi-user kea database is required to manage these data.
- Knowledge of historic range changes since human settlement.
- Knowledge of early Māori traditions, myths, sightings, and interactions with kea.
- An improved understanding of habitat use and frequency/extent of kea movements between different habitats (alpine vs lowland forest, east and west coast, backcountry and human settlements, etc.).
- Knowledge of home range size in different habitats – GPS tracking and the ability to remotely download data will greatly extend the applicability of telemetry for building our understanding of the spatial requirements of kea.
- Knowledge of dispersal and movements of pre-breeding kea.
- Research on the demographics, movements, and behaviour of kea flocks and how these may change seasonally.
- An understanding of the perceived male-bias sex ratio in kea captures and observations.
- Factors triggering breeding, and implications for timing of recovery actions.

3. How to identify and actively manage threats to kea

3.1 Predation by introduced mammals

Early research found no evidence that introduced mammalian predators posed a threat to kea, (Brejaart 1994; Jackson 1960, 1963, 1969). Consequently, early kea management strategies made no mention of managing the threat posed by predators, instead focusing on improving public perceptions of kea and resolving conflicts between kea and humans (Grant 1993; Peat 1995).

The first indications of predator impacts on kea were found between 1992 and 1999 in the St Arnaud Range, Nelson Lakes National Park (Elliott and Kemp 2004). During this period 40 kea nests were monitored and of these, either eggs or nestlings disappeared from 35% of nests, with sign consistent with stoat predation found at two failed nests. The incidence of predation appeared to be relatively low, and population modelling estimates indicated that the kea population was stable at that time (Elliott and Kemp 2004). Nevertheless, these findings showed that kea were vulnerable to predation during nesting. Only stoats were implicated as nest predators during this study, though both stoats and possums were identified as potential nest predators. The kea population in the Nelson Lakes area was resurveyed in 2009–2011, and a 70% decline in the adult kea population was reported since the last surveys in 1999 (Kemp et al. 2022). It is now suspected that this decline was due to extended mast-seedfall-driven predator irruptions recorded at Nelson Lakes and most other eastern beech forests during the interval between censuses (Kemp et al. 2022).

By utilising technological advances in radio and GPS-tracking, remote trail cameras and DNA forensic testing, the causes of kea mortality are now able to be more accurately determined (Kemp et al. 2022). During nesting, stoats are the primary predator of eggs and nestlings, with occasional nest failures also likely attributed to weasels, possums, rats, and feral cats. During carnivore irruption years following beech and/or rimu mast seeding, kea productivity is very low (nesting success c. 10%) in areas without predator control (Kemp et al. 2018; DOC, unpubl. data).

In lowland podocarp forest in South Westland, aerial 1080 improved the odds of daily nest survival by a factor of 9.1 (Kemp et al. 2018). Prior to application of 1080, nest survival rates in a non-treated and treated site were 21% and 47%, respectively. Following the application of 1080, nest survival increased to 85%, whereas it declined to 12% in the untreated site. This positive effect of aerial 1080 on kea nest survival was attributed to the effective control of mammalian nest predators, particularly stoats (Kemp et al. 2018).

Kemp et al. (2022) found that kea of all ages and both sexes are vulnerable to predation by stoats and feral cats whilst roosting and foraging, in certain ecological contexts. This finding was part of a multi-decadal study of annual survivorship of radio-tagged kea. The study includes a sample of kea within dry eastern ecosystems between 2019–2021, upon which a catastrophic episode of kea predation was discovered during a mast-driven irruption of carnivores (Kemp et al. 2022). During this period, survivorship of adult kea in eastern areas was reduced to < 60% per annum due to stoat and feral cat predation (Kemp et al. 2022). This predation episode continued for at least two years, by which time two-thirds of the adults in the population had been killed. Kemp et al. (2022) hypothesise that low survivorship in dry eastern ecosystems resulted from a more pervasive presence of feral cats due to proximity of kea habitat to rabbit infested ecosystems (such as braided riverbeds, tussockland, and farmed grassland/shrubland), and prey switching by both stoats and cats following a sudden crash in mouse abundance. A reduction in adult survivorship is particularly concerning given that population growth rates of long-lived slowly reproducing species like kea require high adult survival rates for population viability (Kemp et al. 2022). However, annual survivorship of kea was reassuringly high in other contexts (adults >90%; juveniles >70%). Both annual survivorship and nesting success are elevated following landscape-scale predator control using aerial 1080 (Kemp et al. 2018; Kemp et al. 2022).

To understand the potential long-term responses of the kea population/sub-populations to predator pressure and predator control programmes, population modelling based on measured survival and recruitment rates at different predator densities is required. Kea inhabit a wide range of ecosystem types across an extensive range. Thus, predator dynamics and kea predation risk are likely to be variable. Future research and management of kea needs to address this spatial variation.

3.2 Lead poisoning

3.2.1 Lead poisoning as a threat

Lead is a highly toxic heavy metal that acts as a nonspecific poison affecting all body systems (Pain et al. 2019), and is a persistent pollutant in the environment. Birds are sensitive to lead exposure, leading to apparent sublethal or lethal toxic responses, and exposure has been documented in more than 120 species (Haig et al. 2014). Once ingested, if lead is absorbed from the gastrointestinal tract it is distributed in the bloodstream and deposited rapidly into the body tissues, especially the liver and kidneys, but also the bones and growing feathers. Lead in the bones is retained for long periods and accumulates through the bird's lifetime whereas lead in the soft tissues and blood is excreted over weeks to months after exposure (Pain et al. 2019).

Multiple studies have described the effects of lead exposure in birds and can briefly be summarised as acute and chronic damage to multiple organs which can result directly in illness and mortality, or subclinical effects which can increase the risk of death due to other causes (Haig et al. 2014). As described in Pain et al. (2019), clinical signs of lead poisoning in birds are often associated with chronic extended exposure at a level that is not initially likely to cause immediate failure of biological function or death, although death may result. Signs include anaemia, lethargy, muscle wastage and loss of fat reserves, green diarrhoea staining the vent, wing droop, loss of balance and coordination, and other neurological signs such as leg paralysis or convulsions. In contrast, after acute exposure to high levels of lead, birds die rapidly without such signs.

In addition to the direct impacts of lead on welfare and survival, indirect effects are likely to occur. These may include increased susceptibility to infectious diseases, parasite infestations (via lead's immunosuppressive effects), and increased susceptibility to death from a range of other causes, via its effects on cognitive ability, muscular strength, and coordination such as collision with power lines and cars, reduced hunting skills, affected flight height, and reduced movement rates (Ecke et al. 2017). Lead can also affect adult reproductive capability, alter growth, development and immune-competence of hatchlings, and reduce early life stage survival (Vallverdú-Coll et al. 2015; Vallverdú-Coll et al. 2016; Williams et al. 2017). The source of lead in the nestlings has been demonstrated in some species to be via transmission in the egg. For example, Wilson et al. (2007) found that blood lead concentrations of female Pacific common eiders (*Somateria mollissima v-nigrum*) increased significantly during incubation. This increase indicated a chronic low-level metabolic release of lead related to reproductive physiology, particularly the mobilisation of stored lead via metabolism of medullary bone (Franson et al. 2000; Wilson et al. 2007). In mallard (*Anas platyrhyncos*) eggs from the

Ebro delta (Spain) eggshell lead and duckling blood lead levels were positively correlated, and ducklings with blood levels > 18 µg/dL had reduced body mass and died during the first week post-hatching (Pain et al. 2019). Nestlings may also develop lead poisoning via foods provided by the parents if the food source is contaminated (McLelland et al. 2010).

At the population level, demographic modelling has shown that lead poisoning is likely to be suppressing the population growth of multiple predatory and scavenging birds worldwide (Finkelstein et al. 2012; Meyer et al. 2022; Slabe et al. 2022).

Blood lead levels of > 40 µg/dL are considered to indicate clinical toxicity in birds, and may be accompanied with clinical signs in some, but not all cases (Platt 2006). Subclinical lead poisoning is considered likely if blood lead levels are within 20–40 µg/dL. Birds with a blood lead level < 20 µg/dL are considered exposed, but less likely to be experiencing toxic effects (Platt 2006). Fallon et al. (2017) recommend similar levels to indicate intoxication, but suggest assessment of the individual and other factors such as breeding status before uplifting wild birds for chelation treatment.

3.2.2 Impacts of lead poisoning on kea

Jackson (1969) first described behaviours and blood results (anaemia) in kea at the Arthur's Pass rubbish dump which are consistent with lead poisoning in parrots. In the same area, Jarrett (1998) detected lead exposure in kea with higher levels in younger birds than adult kea. In 2006–2007, 38 kea were tested for lead toxicity at Aoraki/Mt Cook and all birds tested had detectable blood lead (McLelland et al. 2010). Almost 70% (26 of 38) had levels above 20 µg/dL indicating subclinical lead poisoning and this group included two nestling kea. Additionally, five kea from this area which had died with clinical signs consistent with lead poisoning had detectable lead in either liver or kidney tissue (McLelland et al. 2010). Reid et al. (2012) expanded on McLelland's work to include testing of kea from additional sites and the further testing of stored tissues samples. Across the two studies, a total of 88 blood samples and 20 stored tissue samples from deceased birds were tested from seven locations. All live kea tested had been exposed to lead, and 11 out of 20 tissue samples had elevated liver and/or kidney lead concentrations (Reid et al. 2012).

Treatment of lead poisoning requires veterinary hospitalisation and removal of lead particles via bulk diet therapy, endoscopy, or surgery, combined with chelation and fluid therapy to safely remove the lead from blood for excretion (Richardson 2006). Treatment is generally performed as a series of chelation treatments with rest days in between, during an extended stay in captivity, and blood lead is retested at regular intervals. When the lead is removed from the blood with each chelation treatment, more lead will re-enter from the soft tissues or bones, thus requiring the repeat treatments to effectively remove the majority of lead in the blood and tissues. In the period 2016–2021, at least 25 kea from Arthur's Pass Village were sent for chelation treatment to the South Island Wildlife Hospital (Laura Young, pers. obs.). The DOC-Massey National Wildlife Pathology database contains 189 necropsy reports where a primary cause of death was identified, of which 34 (18%) were reported as lead toxicity based on tissue lead level testing (Appendix) (accessed 23/12/2021, search "kea, 1990 to present", pathologists SA Hunter, MR Alley, BD Gartrell, F Castillo-Alcala, MG Collett, PHG Stockdale, RJ Norman, unpubl. necropsy reports).

Between April 2006 and January 2022, 818 blood samples from wild kea were tested for lead, either as part of direct studies into lead toxicity or incidentally during capture of birds for monitoring, research, or management (KCT and DOC, unpubl. data). Samples were tested using a portable Leadcare® analyser unit (Meridian Bioscience, USA) which enables lead testing in remote locations on a very small (50 µL) blood sample. Of these, 84% had lead above the lower detectable limit indicating lead exposure (> 3.3 µg/dL, n = 689) and almost a quarter (23%, n = 187) of the birds tested had toxic blood lead levels (> 20 µg/dL) (Fig. 1).

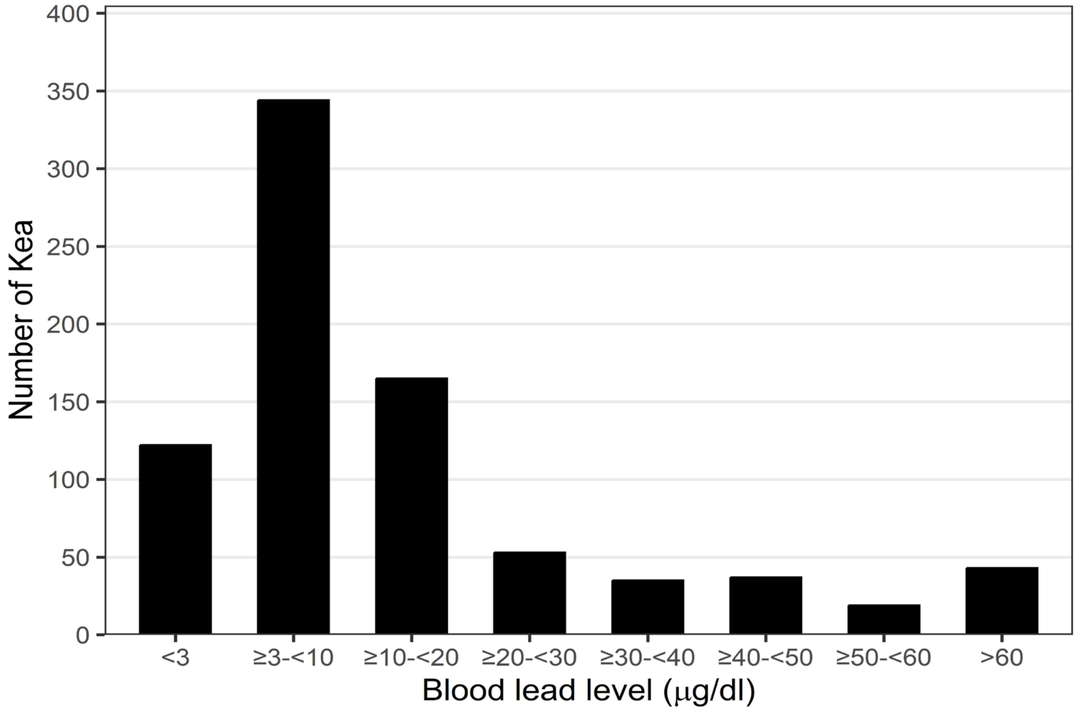

Figure 1. Distribution of blood lead levels of kea sampled from across Te Waipounamu 2006–2022, *n* = 818 (Source data: Kea Conservation Trust and Department of Conservation).

Geographically, higher blood lead levels appear to be clustered around several areas: Central Westland, Aoraki/Mount Cook, and Arthur's Pass Village (Fig. 2). Reid et al. (2012) found that kea living close to permanent human settlements had significantly higher blood lead levels than those birds living remotely, with the majority of kea tested near populated areas having toxic blood lead levels (> 20 μg/dL). This does not explain the high levels of lead found among birds within the more remote areas of Central Westland however (see also section 4.2.4).

An understanding of the effects of lead exposure on kea survival and reproductive capacity is currently lacking, along with predicted impacts on kea population dynamics. Recent research on kea foraging ecology indicates that even low concentrations of lead (< 10 μg/dL) can have sublethal impacts on kea behaviour (McLean 2023). Using stable isotope analysis, McLean (2023) identified an exponential decrease in the trophic feeding level of kea associated with increased blood lead. Individuals with the highest blood lead concentrations consumed mostly plants, whilst those with lower lead levels consumed a higher proportion of invertebrates.

Ongoing blood lead monitoring by researchers, DOC, and community groups will continue to provide additional evidence of continued exposure of kea to lead throughout their range and identify problem areas. However, a nationally structured approach to lead-testing and survival monitoring would improve our understanding of the impacts of lead poisoning on the kea population.

3.2.3 Sources of lead poisoning for kea

Sources of lead within kea habitat include: (1) building materials on huts, facilities, and houses such as lead flashing, lead-head nails and lead paint and contamination of water from lead piping/guttering (Sriram et al. 2018); (2) lead paint, batteries, galvanized wire, and other materials at rubbish dumps (Jarrett 1998); (3) lead wheel weights (Frazer and van der Touw 2014); and (4) ballistic lead from bullets and lead shot pellets used for hunting, either present in scavenged carcasses or found in the environment and eaten (Haig 2014).

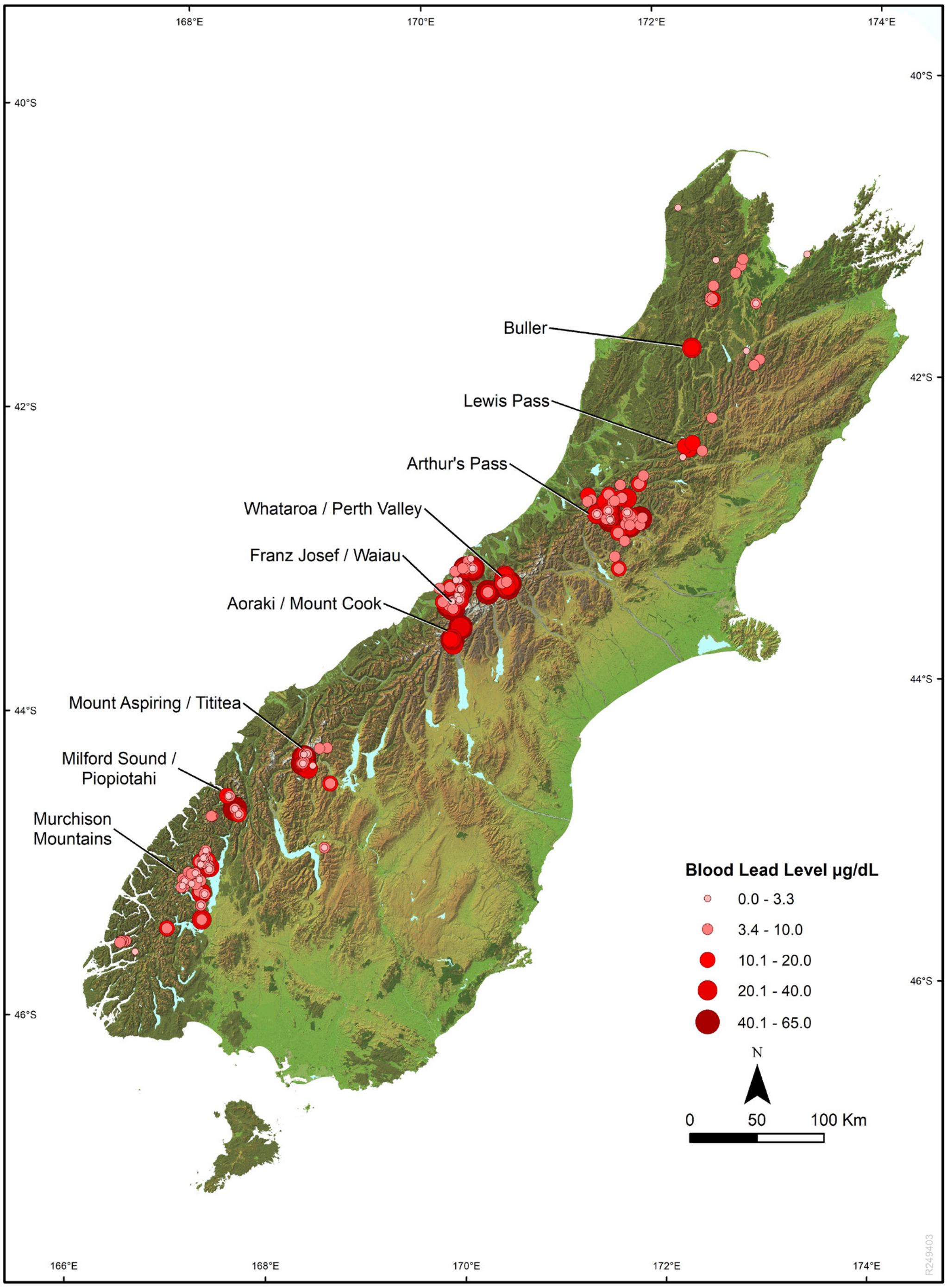

Figure 2. Geographical distribution of kea blood lead-level sampling sites. Larger, darker circles show areas where sampled kea had higher blood lead concentration, *n* = 818 (Source data: Kea Conservation Trust and Department of Conservation).

Isotope testing of lead within blood, feathers, tissue, and bones from live and dead birds is a promising option to identify the primary sources of lead among kea (Church et al. 2006). This technique requires minimal equipment to collect, samples have a long shelf life, and it can provide retrospective data from stored material to build an understanding of lead exposure history (Finkelstein et al. 2012).

Kea, particularly fledglings and juveniles, are known for their inquisitive and destructive behaviour and are attracted to lead possibly due to its malleable nature and sweet taste (Alley 2002; Reid 2008). Exposure to human foods, either by deliberate feeding or accidentally through access to waste food, provides an easy energy source, which allows kea to abandon natural foraging activities. This enables kea more time to be exposed to lead sources when, for example, chewing on buildings or vehicle wheel weights (Diamond and Bond 1999; Reid 2019).

Removal of lead from infrastructure at lead hotspots was initiated through the KCTs Jobs for Nature-funded lead removal project at the end of 2021. A number of sites (e.g. Arthur's Pass and Aoraki/Mt Cook villages) are set to have the roofs of all buildings completely lead-free by the end of 2023 (KCT 2023). An ongoing lead removal programme at lead hotspots would reduce the incidence of kea accessing and ingesting lead.

3.2.4 Implications of wild animal control and hunting using lead ammunition

One source of lead poisoning of birds is ammunition, resulting from the ingestion of lead bullet fragments directly, or from feeding on the carcasses of animals shot with lead ammunition. Lead bullets expand and then fragment whilst dissipating the bullet's energy on impact. As a result, hundreds of lead fragments can remain in the carcass, particularly around the point of entry, where scavenging birds are likely to feed. This fragmentation increases the likelihood that multiple individuals can be exposed to lead from a single carcass. Negative effects are also increased when carcasses are used as food resources by long-lived, widely dispersing, social species which feed communally (Haig et al. 2014).

Lead ammunition has been banned for waterfowl hunting in many parts of the world for decades, including the United States (since 1991) and many parts of Europe (including Denmark in 1986). New Zealand followed more recently, when the completion of a staged approach in 2021 led to a complete ban of all but 410-gauge lead shot for waterfowl (within 200 m of a water body). However, lead exposure remains a problem for many avian species on land. The relationship between elevated lead levels or exposure rates in avian scavengers and hunting activities on larger game animals has been observed in multiple species across the world (Haig et al. 2014). In the United States, lead poisoning attributed to lead ammunition is a leading cause of death among California condors (*Gymnogyps californianus*), and thought to be the primary factor preventing their population recovery (Finkelstein et al. 2012). In Japan, lead poisoning is prevalent among Steller's sea eagles (*Haliaeetus pelagicus*) and white-tailed sea eagles (*H. albicilla*) for whom the carcasses of sika deer are a major food source over winter (Saito 2009). Across Europe, lead poisoning has been described in 17 species of birds including the threatened Spanish imperial eagle (*Aquila adalberti*), golden eagle (*A. chrysaetos*), and four species of vulture (Mateo 2009). More recently, lead isotope analysis to identify the sources of lead in the environment has led to the confirmation of lead ammunition as the primary source of lead poisoning for many of these species (Finkelstein et al. 2012; Ishii et al. 2020; Madry et al. 2015).

Kea are known to possess many of the traits putting them at very high risk of lead poisoning via consumption of contaminated carcasses. These traits include being a long-lived, social, widely dispersing species that scavenges communally. A necropsy of a juvenile male kea confirmed to have died of lead toxicosis revealed a partially digested lead shot pellet in the stomach (R. Norman, unpubl. necropsy report). Wild animal carcasses such as tahr (*Hemitragus jemlahicus*), deer (*Cervus* spp.), and chamois (*Rupicapra rupicapra* subsp. *rupicapra*) appear to be an important food source for kea in alpine areas particularly, and flocks of kea are frequently observed scavenging together on a single animal carcass (Brejaart 1988; Nichols and Bell 2019; Schwing 2010).

Large numbers of game animals are regularly culled through animal control programmes within kea habitat each year, as part of the Wild Animal Control Act 1977, with thousands of carcasses left to decompose naturally. Prior to 2019, lead ammunition was routinely used throughout these programmes. However, in 2018, high levels of lead were detected in blood samples from kea sampled in the remote Adams Wilderness Area and it was suggested that kea scavenging the carcasses of tahr controlled with lead ammunition may have been a significant contributor to those test results (Tracey Dearlove, pers. obs.). A decision was made to switch to lead-free shotgun ammunition within DOC's tahr control programme, which involves culling several thousand tahr each winter in kea habitat as part of the Himalayan Thar (tahr) Control Plan (1993). Safety concerns around a potential bullet ricochet event resulted in DOC reverting to lead-ammunition between July and November 2019 when c. 8,560 tahr were dispatched and approximately 405–459 kilograms of lead shot was discharged into the environment. However, following a formal safety investigation, DOC recommenced using lead-free shotgun ammunition for tahr control from July 2021 under the 2021/22 Tahr Control Operational Plan and has used lead-free shotgun ammunition thereafter. A very limited number of tahr were controlled using lead-core centrefire ammunition in 2021/22 as the programme continued its progress in also phasing lead out of centrefire ammunition. Substantial trialling of lead-free centrefire ammunition for aerial and ground-based tahr control has been undertaken since 2019. Commencing with the 2022/23 Tahr Control Operational Plan (July 2022), the tahr programme will utilise 100% lead-free ammunition for all aerial- and ground-based operations; a shift likely to see a substantial reduction in risk to kea of lead poisoning via this pathway. The risk from other wild animal control programmes within kea habitat has yet to be assessed. Recreational hunting within kea habitat also presents a significant risk to kea, particularly during concentrated periods of activity such as during the annual winter tahr ballot in the central Southern Alps and South Westland, when food is otherwise limited for kea across the landscape.

Analysis to identify the primary sources of lead toxicity in kea using lead isotopes has not been conducted for kea. However, a recently established research collaboration between DOC, Te Rūnanga o Ngāi Tahu (TRONT), Nelson Marlborough Institute of Technology (NMIT), South Island Wildlife Hospital, University of Melbourne, and University of California, Santa Cruz is exploring whether this is a viable option.

3.3 Non-target impacts during pest control operations

All predator control tools present some level of risk to non-target species. The exploratory behaviour of kea means that they are prone to accidental death or injury in predator control operations.

Despite the proven benefits of aerial 1080 in boosting kea productivity and survivorship (see section 3.1 above), intensive radio-tracking of individuals through aerial 1080 operations has shown that kea can be at risk of primary poisoning (also see section 4.1) (Kemp et al. 2019). Research over the past decade has identified several potential factors as influencing non-target risk to kea such as pre-feeding and bait sowing rates (DOC 2020), composition of the bait matrix (Blyth 2011), bait additives (McLean et al. 2022a), bait colour and lure type (Brunton-Martin et al. 2021; Cowan and Crowell 2017; Grosser et al. 2022; Weser and Ross 2013), previous exposure to 1080, and proximity of the control operation to human habitation (Kemp et al. 2019). This research forms the basis of the Department of Conservation's Aerial 1080 in kea habitat Code of Practice (COP) (DOC 2020). The Code of Practice is the primary guide in managing risk to kea through aerial 1080 operations based on the best available research to date. The COP is regularly reviewed to ensure currency of knowledge around predator control and kea ecology.

The most recent major revision of the COP recognises that kea mortality is not evenly spread across all 1080 operations and applies compulsory performance standards based on this variation in risk (DOC 2020). These risk factors were identified by Kemp et al. (2019), who analysed the survival outcomes of 222 radio-tagged kea through 19 predator control operations using 1080 between 2008 and 2016 (Kemp et al. 2019). They investigated the risk of 1080 poisoning with respect to kea age and sex, 1080 bait size, 1080 treatment history, and proximity of the site to human habitation. Two of these variables (the history of 1080 treatment at a site, and proximity of the control operation to human habitation) were found to have a significant effect on kea survival. Specifically, kea were less likely to survive a 1080 operation if they lived less than 20 km from places of human habitation where kea congregate and obtain human food sources, sites commonly referred to as 'scrounging sites' (Kemp et al. 2019). This may be because young kea exposed to human-made objects/food at scrounging sites are likely to have an expanded array of what they will explore/consider as food (also see Section 4.1) (Diamond and Bond 1999). Additional factors may also increase poisoning risk to kea near scrounging sites, such as lead poisoning and social facilitation, whereby a heightened risk of mortality that more explorative kea face spreads to other individuals that observe them feeding on baits (Diamond and Bond 1991; Gajdon et al. 2006; Reid 2019). Kea survival was also found to be lower during aerial 1080 operations where the site had not previously had 1080 applied (Kemp et al. 2019). The mechanism for this effect is also unclear, but may involve bait-shyness via sublethal poisoning (conditioned aversion), or selection pressure against bolder, more explorative individuals (Kemp et al. 2019).

Aside from attempts to reduce scrounging sources for kea (see Section 4.1), additional 1080 risk mitigation work is focused on the development of chemical repellents that can be added to baits to deter kea (Cowan et al. 2016; Nichols et al. 2020; Orr-Walker et al. 2012; Weston et al. 2021). Recent work using the secondary repellent anthraquinone, which causes gastrointestinal discomfort and an emetic (i.e. vomiting) response when consumed, has shown promising results during captive trials (McLean et al. 2022b; Nichols et al. 2020) and subsequent field operations (Nichols and Bell 2019). However, anthraquinone cannot be freely applied to baits within a treatment area given that it also has a repellent effect on target pests (i.e. rats) (Cowan et al. 2015; Nugent et al. 2020). Diversionary feeding using other potential food sources such as wild animal carcasses is also currently being explored as a potential risk mitigation method for kea during aerial 1080 operations (Nichols and Bell 2019).

Non-target impacts to kea have been recorded when using a range of ground-based pest control devices, and few kea-safe devices are currently available for deployment in kea habitat (KCT 2022). Enclosing traps and poisons within 'kea-proof' boxes and stations has mitigated some risk. However, these measures can fail due to the unique problem-solving abilities of kea. For example, kea have been found to be able to access DOC 150/200 traps by removing the screws or nails holding down the lids. Between 2008 and 2021, 50 kea deaths have been recorded in 10 trap types in mustelid, feral cat, or possum trapping operations (DOC, unpubl. data).

Kea will access chicken eggs used as lures in traps and have been observed eating the carcasses of both stoats and rats caught in DOC 200 traps (Jamie McAulay, pers. obs.). This food reward likely drives further exploration of trap boxes. Kea deaths in stoat traps are spatially and temporally clustered, with an apparent social learning pathway of kea developing various techniques, including tool use, to access protein sources from traps (Goodman et al. 2018). Kea open traps and are killed by snipping through wire trap ends, digging out lid screws, tumbling trap boxes down hills, and digging through the wooden sides of trap boxes. Efforts to deter kea access, including stainless trap ends, additional lid screws and securing trap boxes have proven effective at deterring this behavior, but work best before a food association with trap boxes is learned (Goodman et al. 2018). Trials applying an anthraquinone-based repellent formulation (Arkion® Life Sciences, AV-4044) mixed with tree paint to trap boxes have recently begun in Fiordland. Early indications are that the repellent paint mix has reduced kea interactions with the trap boxes, but this approach requires further testing (Andrew Smart, DOC, pers. comm.).

One kea death has been recorded during a cyanide bait station operation, and kea interference with cyanide bait bag operations has also been reported (Fairweather 2018). A kea death has also been recorded during a 1080 bait station operation (Poutu et al. 2021). While kea deaths from other toxins within bait stations have not been recorded, it is expected that any of the various baits used for vertebrate pest control would potentially place kea at some risk, unless used in a bait station that prevents kea access (Nic Gorman, DOC, pers. comm.).

DOC manages any identified risks with pesticides and traps with performance standards for each pest control tool, which are to be followed whenever these tools are used on Public Conservation Land. In many cases, kea are specifically addressed in these standards, ranging from prohibition of use of the tool in kea habitat, restrictions on types of lures, and where, when and how tools can be used. Any evidence of kea interaction with a pest control device is recorded. These standards are based on current knowledge (which is often limited, particularly with newer tools) and are intended to be revised in light of new findings. A best practice guide for safe pest control in kea habitat that outlines the risks associated with various pest control devices and methods to minimise these risks has also been developed by the Kea Conservation Trust (KCT 2022).

3.4 Avian diseases

Health data on kea has been reviewed from multiple sources including clinical, diagnostic, and necropsy results on captive kea both in New Zealand and overseas, and anecdotal reports, limited surveillance studies, and necropsies of wild kea. To the best of our knowledge, deliberate transmission trials to test susceptibility to avian pathogens have not been undertaken in kea.

There are several parrot-specific diseases that kea would be potentially susceptible to if they appeared in New Zealand (e.g., bornavirus [proventricular dilatation disease], psittacine beak and feather disease virus [BFDv], polyoma virus, avian herpesvirus [Pacheco's disease], and psittacine avipoxvirus). However, levels of infection across parrot species tend to be variable, ranging from asymptomatic through to morbidity and mortality. A good example is BFDv, which causes little or no symptoms in budgerigars (*Melopsittacus undulatus*) but results in feather loss, beak deformity, and immunosuppression in sulphur-crested cockatoos (*Cacatua galerita*). For this reason, it is not possible to predict with any certainty what effects parrot diseases will have on kea. There is no evidence that kea possess an innate immunity to the suite of diseases generally associated with parrots and other avian species. However, data are limited due to the small numbers of kea held in captivity, limited access to appropriate testing, and the low level of reporting of kea health data in the international literature. Additionally, extrapolation of health data from captive to wild birds may not accurately reflect exposure and susceptibility of kea to these pathogens.

Many of the major parrot diseases of aviary flocks internationally have not been detected in New Zealand, which can be attributed to the ban on importation of live birds since 1997. Psittacine avipoxvirus (Gartrell et al. 2003), Pacheco's disease (Durham et al. 1977; Thornton and Stanislawek 2003; Valastanova et al. 2021) and psittacine BFDv (Massaro et al. 2012) have all been detected in New Zealand among exotic species held in captivity, though these have not been detected in kea to date. Of these three diseases, only one has been the subject of disease survey among native parrots: in 2010, 95 wild kea from across their range and five captive kea (two from Te Ika-a-Māui , three from Te Waipounamu) were tested via PCR of blood (98) or feather (2) samples for BFDv with no positive results detected (Massaro et al. 2012). Continued careful management of importation risks are needed to prevent these diseases or novel genotypes of these diseases from reaching New Zealand and potentially spreading into wild populations of parrots (MPI 2012).

The DOC-Massey National Wildlife Pathology database currently contains 224 kea necropsy reports from 1990 to 2021, of which 189 had a confirmed primary cause of death reported (accessed 23/12/2021, search "kea, 1990 to present", pathologists S.A. Hunter, M.R. Alley, B.D. Gartrell, F. Castillo-Alcala, M.G. Collett, P.H.G. Stockdale, R.J. Norman, unpubl. necropsy reports) (Appendix). Fifty-four kea were from captive institutions and 170 were found dead in the wild, or were uplifted as a sick bird and died in care. Not all birds that are found dead are submitted for necropsy, therefore this is not a comprehensive census of kea deaths. However, the database provides a passive surveillance method for detecting disease. Necropsy does not always determine a definitive diagnosis of cause of death and some deaths may have multiple causes (e.g., infection could cause a bird to be weak and thus more vulnerable to being killed by a predator). In this review, the primary diagnosis was used to categorise the most likely cause of death. If no clear cause of death could be determined from the necropsy, then the diagnosis is defined as 'open'. Most kea necropsied at Massey University died from toxicity or trauma (69%), with infection the third most significant category (Fig. 3). No kea sent for necropsy, wild or captive, was diagnosed as having died of an infectious avian disease.

When this information is refined based on the origin of the kea submitted, it reflects the difference in the environmental conditions of wild and captive kea, including the degree of human observation, interaction, and intervention (Fig. 3). Wild birds are more likely to be found if the bird dies near human habitation or is part of a study and wearing a radio-transmitter, and the majority of diagnoses for wild kea reflect an acute cause of death – by toxin (e.g., lead, 1080, chocolate) or trauma (e.g., vehicle strike, predation, trap), with infection as the third most common cause. Captive kea submitted for necropsy are usually adult birds (95% of submissions), with the cause of death ranging broadly across categories. Captive kea are more commonly reported with diseases of old age such as neoplasia (cancer), arthritis, and chronic respiratory infections. This does not appear to be a species-related infectious disease susceptivity, rather it is a function of the extended lifespan provided by the captive

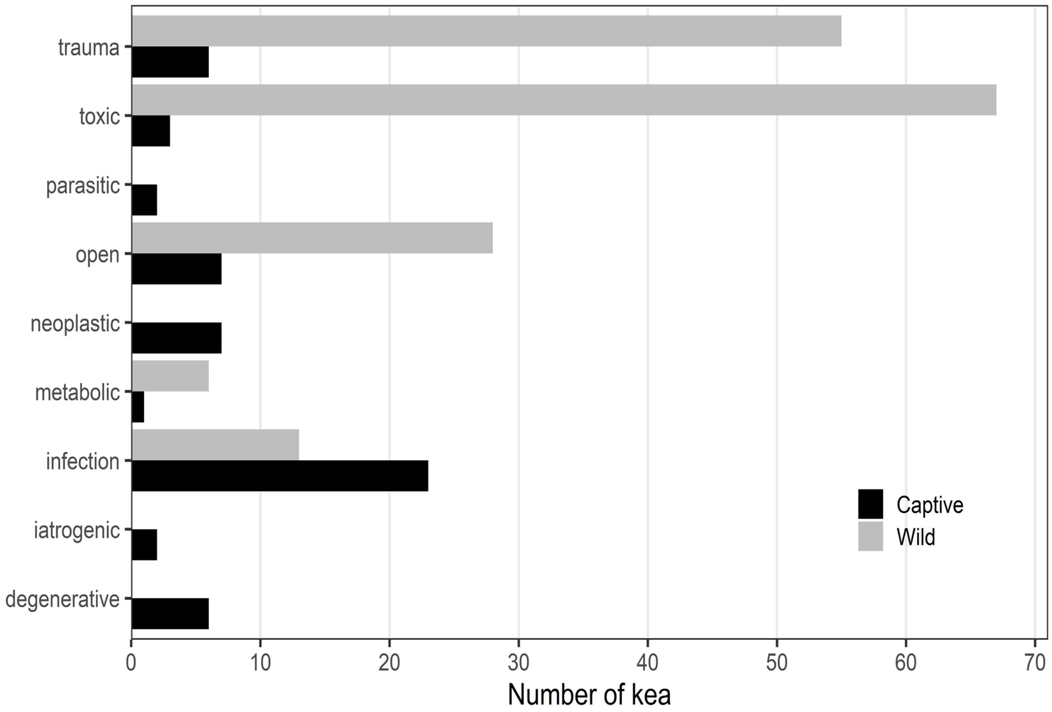

Figure 3. Origin of kea (wild or captive) and cause of death recorded in the DOC-Massey National Wildlife Pathology database 1990–2021. Diagnosis categories: 'trauma' – damage to tissues and structures caused by force; 'toxic' – the presence of toxin and/or tissue damage attributable to toxic effects; 'parasitic' – presence of parasites causing disease; 'open' – no clear cause of death could be determined from the necropsy; 'neoplastic' – presence of cancerous cells; 'metabolic' – disruption or decline of normal biochemical pathway in the body; 'infection' – presence of, or damage caused by pathogens (germs); 'iatrogenic' – damage due to medical/human intervention during treatment of disease; 'degenerative' – the slow degrading of body structure and function due to age (source: Department of Conservation/Massey University).

environment and intensive veterinary diagnostics of captive kea. The largest number of reports were in the 'infection' diagnosis category, with bacterial and fungal infections across a range of body tissues such as lung, liver, kidneys, eye, and reproductive tract. Similar infections in wild birds are only likely to be detected if the bird is in regular contact with humans.

Potential susceptibility of kea to exotic avian diseases can be perhaps best indicated by records of disease incidence in kea from captive facilities overseas. For example, psittacosis (*Chlamydia psittaci*) is globally distributed and commonly detected in avian species, causing respiratory disease, septicaemia, and occasionally death (Gedye et al. 2018). This disease was introduced to New Zealand via imported parrots from Australia (Cairney 1954) and a recent outbreak in the Auckland region caused deaths of several hundred wild Malay spotted doves (*Streptopelia chinensis*). It also infected humans who had contact with dead birds (Rawdon et al. 2009). However, the only record of psittacosis in kea is from an overseas report by Bell and Schroeder (1986), who described incidence among healthy kea that were imported to a captive collection in the United Kingdom. Similarly, Sandmeier et al. (1999) described the detection of avian polyomavirus (APV) in a recently acquired kea at a captive facility in Switzerland. In New Zealand, avian polyomavirus had previously been detected in a fatal outbreak among captive Gouldian finches (*Chloebia gouldiae*) (Alley et al. 2013) and it is considered endemic within captive budgerigar (*Melopsittacus undulatus*) flocks (Baron et al. 2014).

Detection of the avian influenza (AI) virus in a dead kea held at the National Zoological Gardens of South Africa indicates a potential susceptibility of kea to this disease for which non-symptomatic carriers are common (Burger et al. 2010). Regular surveillance for AI in mallard ducks (*Anas platyrhynchos*) and migratory shorebirds in New Zealand is undertaken by the Ministry for Primary Industries to monitor circulating non-pathogenic strains and detect arrival of pathogenic strains (Langstaff et al. 2009; Stanislawek et al. 2002). West Nile Virus (WNV), which is spread by mosquitoes and some other biting insects, has been reported in a wide range of species overseas including captive parrots (e.g. https://www.cdc.gov/westnile/resources/pdfs/BirdSpecies1999-2016.pdf). Although WNV has not been recorded in New Zealand, Bakonyi et al. (2016) reported an infection of six kea in a captive institution in Vienna with WNV in 2008, which caused mild to fatal neurological disease.

Parasites can contribute to death in kea, though in general there is usually an underlying stressor such as injury, toxicity, lack of nutrition, or behavioural abnormalities, which suppress the immune system and allow parasite multiplication. Parasite load as a primary cause of death is only likely where parasite exposure is high (e.g., in a captive situation where continual reinfection via ingestion of eggs or oocysts results in an artificially high burden). Five species of helminth (worm) have been associated with kea (McKenna 2010).

Researchers and field staff play an important role in the monitoring of kea for health and disease; baseline health information should be collected when possible. An expansion of disease surveillance to include regular targeted testing for diseases of interest in kea would be beneficial. Reporting, and investigation of sick or dead kea, and maintenance of international biosecurity to prevent arrival of exotic diseases and novel disease genotypes should also continue.

3.5 Climate change

New Zealand's terrestrial biodiversity is under increasing pressure as a result of global climate change (Christie 2014). The impacts of climate change on native biodiversity can potentially be either direct (such as changes in range, seasonality, and abundance) or indirect (exacerbating other threats such as pests and human disturbance) (Macinnis-Ng et al. 2021).

For example, exposure to pests, pathogens and diseases not currently encountered by kea, or an increase in exposure rates to those within range, may be precipitated by climate change.

The potential for increased predator pressure is of particular concern regarding kea (Christie 2014; Kemp et al. 2022). Rat abundance is linked to higher temperatures, in that warmer overall temperatures may lead to fewer periods of naturally low rodent abundance during winter in montane forests and alpine ecosystems. More rodents can be expected to translate to more invasive carnivores (Christie et al. 2017; Studholme 2000). However, complex interactive effects between resources, productivity, temperature, and guild assemblage makes predicting future impacts of pests under climate scenarios challenging (Macinnis-Ng et al. 2021). Recent research investigated the likely response of endemic New Zealand forest birds to range expansion of mammalian predators by looking at drivers of recent decline (Walker et al. 2019). Species that are large, nest in tree cavities, and/or disperse poorly were found to have undergone more rapid recent loss where temperatures are higher, consistent with higher and more constant predation pressure in warmer forested sites. Large, strongly-dispersing, cavity-nesting species, such as kea, did not undergo as rapid a decline, but their local occupancy decreased significantly with increased forest temperature (Walker et al. 2019).

The elevational range shifts that will occur as a direct consequence of climate warming are expected to have serious consequences for species utilising alpine habitats (Chamberlain et al. 2013). In New Zealand, a loss of alpine habitat as the treeline rises is predicted to have a substantial impact on kea, given that alpine areas comprise important feeding habitats (McGlone and Walker 2011). Increasing fragmentation of habitat may also impact kea by exacerbating edge effects and increasing predator pressure within remaining forest fragments (Ewers and Didham 2006).

An increasingly drier climate, as projected for eastern regions, is likely to increase drought severity and fire risk within kea habitat. In contrast, increased rainfall and storm events, particularly in southern and western areas, may impact habitat through flooding, wind damage, erosion, and landslides (Bodeker et al. 2022).

Other possible impacts of climate change on kea, in relation to their reproductive biology, food availability, behaviour, or interactions with humans long-term, are unknown.

3.6 Illicit trade in wildlife

Kea are absolutely protected under the Wildlife Act 1953. They cannot be collected from the wild, killed, disturbed, possessed (live or dead), held in captivity, sold, or exported without proper authorisation. As in most parts of the world, the illegal take and trade in wild animals is a significant issue in New Zealand. Species which are of value to smugglers include our rare parrots, reptiles, and plants as these are either relatively easy to conceal and/or particularly lucrative on the international wildlife market.

New Zealand has a strong commitment to preventing, investigating, and prosecuting illegal wildlife trade as a Party to the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). The Wildlife (Smuggling Deterrence) Amendment Bill, passed in 2012, provides an increased deterrent to would-be smugglers by increasing the maximum penalty for smuggling native animals from six months imprisonment or a $100,000 fine to up to five years imprisonment and/or a $300,000 fine. Monitoring is maintained by Biosecurity New Zealand at the border.

Recorded incidents of kea smuggling are rare, although this does not mean they do not occur. Kea are curious about human activity and young naïve birds and habituated adults can be relatively easy to catch. In 1990, two suitcases confiscated at Christchurch International Airport, bound for Singapore, contained eight drugged kea stuffed into plastic tubes, one of which was already dead. The kea had been stolen from three zoos and two National Parks in Te Waipounamu (Diamond and Bond 1999).

Prior to becoming fully protected under the Wildlife Act (in 1986), hundreds of kea were legally exported from New Zealand to zoos and collectors around the world. While the origins of these birds were not recorded, it is likely that most were wild caught. Since 1996, no live kea have been legally exported from New Zealand. The CITES trade database (https://trade.cites.org/en/cites_trade) shows 222 live kea were traded internationally between 1981 and 2022, mostly from captive breeding facilities, with a third of these being for personal or commercial purposes. The number of kea held overseas is not accurately known, because some kea are kept outside of coordinated captive breeding programmes. It was hoped that the increasing availability of captive-bred kea originating within Europe and the USA would lead to a decline in the monetary value of kea and consequently reduce the incentives to trade illegally in this species (Puller 1996). However, reports of overseas pet stores selling kea for large sums of money from legally imported stock remain relatively common, such as one UK store selling kea for 3,000 GBP per bird (DOC, unpubl. data) and a café in Japan offering a 3-year-old male kea for ¥1,800,000 ($25,524 NZD) (DOC, unpubl. data). Increasing public awareness of the potential risk to kea of wildlife smuggling, and encouraging local communities to report any suspicious human behaviour in kea habitat, should be encouraged.

3.7 Managing threats – knowledge gaps

Based on the review above, the following are gaps in our knowledge that require further research and tool development to assist in recovery actions for kea:

- An improved understanding of how predator guilds and their impacts differ among kea habitats/sub-populations.
- An understanding of how, when, and where to control feral cats to minimise impacts on kea.
- Measurement of the long-term population benefits of predator control for kea across their entire range, including spatially explicit population modelling.
- An understanding of the extent and impacts of lead poisoning on kea survival, reproductive capacity and the long-term viability of kea populations.
- Research on kea-safe pest control devices, particularly for targeting possums and feral cats. Active encouragement of tool development is required, and all reasonable assistance should be given to developers to hasten testing of new kea-safe devices.
- Review of pest control methods and impacts on kea, including collation of 1080 data.
- Research on effective kea repellents, especially those which can be deployed within toxic baits and effectively delivered in all habitat types.
- Research on the effect of diversionary feeding on mitigating non-target 1080 impacts on kea.
- Development of robust monitoring tools to measure the effectiveness of risk mitigation techniques.
- Carry out lead-source analysis using lead isotopes to ensure the most important lead sources are being targeted by management.
- Analysis of the current kea blood-lead dataset to further identify risk factors and problem areas for lead poisoning.
- Development of a multi-user kea database to hold results of all health testing and treatment of kea from veterinary clinics, DOC, and KCT.
- An understanding of the likely direct or indirect impacts of climate change on kea.

4. Living in harmony with humans

4.1 Managing kea-human interactions (scrounging)

Early management strategies developed for kea identified the risks associated with kea-human interactions. One of the key themes of the 'Wild Kea Management Statement' developed by DOC in the early 1990s was *'to avoid providing kea with supplementary food (intentionally or unintentionally) and discourage them from congregating in areas of human use'* (Grant 1993, p 13). Recent research has highlighted the human influence on the ongoing decline of the kea population. As a result of close association with humans and their property, kea are at an increased risk of poisoning by 1080 (Kemp et al. 2019; Orr-Walker et al. 2012), lead (Reid et al. 2012), and certain foods (Gartrell and Reid 2007). More recently, it is thought that this increased risk may also be the result of learned neophilic behaviours (e.g., seeking new food sources, strong affinity for anything new) through kea obtaining human food by stealing, soliciting, and scavenging (termed scrounging) either directly from people (e.g., a tourist locations) or from rubbish bins and landfills (Kemp et al. 2019).

Recent research on the spatio-temporal habits of kea at one visitor-dense area of kea habitat within Arthur's Pass National Park found that kea (initially captured for monitoring in human areas) strongly selected human areas, especially during periods when human activity was high (Kennedy 2017). Kea at this locality also spent more time on ground-based activities than in flight, thereby increasing the likelihood of interacting with humans.

Although the effect of scrounging has not been studied directly, kea who live near high visitor-density locations have increased opportunity to scrounge (Kennedy 2017). This is thought to lead to an increase in neophilic behaviours, which then has extended consequences in first-time aerial pest operations where kea become interested in 1080 baits. Kemp et al. (2019) found that kea that are located near to these scrounging sites (< 20km) have significantly decreased survival odds (6.9 times lower) during predator control operations compared to kea that reside further away (> 40km) (Kemp et al. 2019).

While mitigation of risk for unintentional kea poisoning has focused on technical aspects of predator control operations (DOC 2020), there has also been a recent focus to understand the human dimensions associated with kea scrounging, such as peoples' tendency to feed and/or interact with kea. This includes a recent literature review highlighting the complexities of the challenges associated with deterring visitors from interacting with kea (Aley 2021). Two main areas of influence are suggested: wildlife tourism practices, and human-avian history and culture.

4.1.1 Wildlife tourism

Human-kea interactions usually stem from chance encounters, that often progress to more sought-after interactions once kea are present (i.e., taking photos), which can result, albeit less often, in offering of food or novel items (Aley 2020b). Underlying these behaviours is the intangible influence of psychological benefits to the person involved, whereby they subconsciously seek a connection with nature – termed the 'biophilia hypothesis' (Wilson 1984). Curtin (2009) found that human-wildlife encounters can result in a 'deep sense of wellbeing that transcends the initial encounter' (p. 451). This, associated with the upward trend of people travelling to connect with nature (Curtin 2009), has resulted in the extension of wildlife feeding as a way of increasing the likelihood of sighting and/or engaging with wildlife (Newsome et al. 2005; Orams 2002). Complicating these encounters can be the intentional discord between an individual's awareness of the unethical implications of feeding wildlife and their actions of doing so (Sezer et al. 2015; Tenbrunsel et al. 2010). This is termed an 'ethical blind spot' and overrides what is considered acceptable in the moment (i.e., they offer food to kea despite understanding feeding wildlife is not acceptable).

Further complicating drivers for human-kea interactions is the role that the capturing and sharing of wildlife images plays in wildlife tourism experiences (Pagel et al. 2020). Sharing images on social media acts like a positive feedback loop, with increases in social media posts showing human-wildlife interactions establishing increased wildlife tourism expectations (Pagel et al. 2020). This trend of 'wildlife selfies' has led to Bell and Lyall (2002) highlighting that travellers have transitioned from 'just looking' to using wildlife as 'actors or props' when accumulating travel experiences, and particularly so for encounters in natural environments (Cong et al. 2014).

4.1.2 Human-avian history and culture

Human interaction with birds is a near-universal practice throughout the world (Jones 2018), with birds being the most popular of all wildlife for people to feed (Orams 2002). Participation rates of households indicate 46% of New Zealanders (Galbraith et al. 2014), 63% of Australians, and 34–75% in the USA and UK feed wild birds (Jones 2018). The influence of this household feeding on the birds social learning is also likely to influence their behavioural interactions with humans through the reinforcement of the provision of food (Donaldson et al. 2012). For example, for urban bird communities, Plummer et al. (2019) found that bird density increased in gardens with feeders, including for birds that had rarely been seen there before.

Underlying human motivations to feed birds are associated with the social drive of all cultures to share food (Orams 2002), the need for people to connect with nature (Cox and Gaston 2016; Jones 2018), and moralistic drivers to care for wildlife as a conservation activity (Jones 2018). Additionally, the human-parrot bond spans thousands of years with a long history of anthropomorphising pet parrots (Boehrer 2015). It is suggested this strong bond is linked to parrots' intelligence, ability to mimic speech, social nature, and longevity (Boehrer 2015) and how these qualities blur the boundary between them and humans (Bond and Diamond 2019; Duncan and Hawkins 2010).

To better understand the current human behaviours at high human-kea interaction locations, an ethnographic study was undertaken recently (Aley 2020b). While the prevalence of observed direct feeding of kea was low (4%), anecdotal reports of people working at the locations suggest it is more common than this. Additionally, the influence of people preparing and eating food outside of their vehicles is a likely unintentional scrounging source for kea, either while the visitor is still present or once they have departed. The study highlighted that kea only initiated interactions with people when they were eating, when novel objects were held out to them, or by flying onto vehicles. Additionally, people were observed to persistently encroach into spaces kea had retreated to, even when they were already within 1–3 metres of the kea. This indicates that although kea are social and 'show high levels of curiosity, object exploration and manipulation and foraging' (Huber and Gajdon 2006; p. 295) they are very selective about the circumstances in which they engage with humans.

Acknowledging the above complexities of human-kea interactions, DOC is currently focusing social science research on shaping appropriate human behaviours when interacting with kea. The objective of this research is to prioritise, design, and test interventions to reduce scrounging sources for kea (Aley 2020a).

4.2 Human-induced injury and death

Kea are inquisitive and are attracted to new objects in their environment (neophilic) (Diamond and Bond 1999). As a result, some tend to interact with humans, their resources, and infrastructure, which is often to their detriment. Aside from the non-target impacts of predator control, and lead poisoning (see sections 3.2 and 3.3), kea continue to be found dead or injured as a result of illegal shooting, blunt force trauma, vehicle strike, interaction with high voltage cables, and interactions with other foreign objects (e.g., rubber, wires, string).

The majority of the 189 kea necropsied at Massey University between 1990–2021 (for which a primary cause of death was determined) died from toxicity (38%) or trauma (32%) (accessed 23/12/2021, search "kea, 1990 to present", SA Hunter, MR Alley, BD Gartrell, F Castillo-Alcala, MG Collett, PHG Stockdale, RJ Norman, unpubl. necropsy reports). The majority of toxins identified were either 1080 (n = 28) or lead (n = 34), although chocolate toxicity (n = 2) and other unknown substances were also recorded (n = 7) (Appendix).

Within the trauma category, the most common cause was vehicle strike (n = 18), followed by ballistic (n = 11) and a range of other blunt-force injuries. Vehicle strike is a particular problem where state highways pass through high elevation settlements such as Arthur's Pass Village, where 12 kea were killed on the road between 2020 and 2022 (Kerry Weston, pers. obs). Electrocution is also a known cause of death for kea. During just one fortnight in 2013, five kea were electrocuted at an electricity substation located behind Unwin Lodge at the entrance to Aoraki/Mount Cook National Park. When asked, Alpine Energy staff said dead kea had been found in the substation in the past. Initial attempts were made to discourage kea from the site by spraying repellent around the substation, however modifications have subsequently been made to eliminate risk to kea and no further deaths have been reported (Dean Nelson, DOC, pers. comm.). In 2017, two kea were electrocuted at the Temple Basin Ski Area in Arthur's Pass. Temple Basin Ski Field and Orion subsequently fixed the cables that kea had chewed on to reduce the risk of electrocution (Tamsin Orr-Walker, pers. obs.).

Aside from the ongoing management of risk to kea at sites of human activity, such as ski fields and alpine villages, identification of other 'at risk' areas is a priority. Erection of signage in these areas to alert the public of potential threats to kea as well as work to educate and support communities should also be undertaken.

4.3 Human-kea conflict

Kea have a long history of persecution in New Zealand, largely stemming from conflict with humans. Kea gained absolute protection in 1986, which encouraged a shift in public attitude from persecution to protection, which was reinforced by early conservation efforts to promote a mindset of learning to live with kea (Grant 1993; Peat 1995). However, some land users and visitors in kea habitat still consider kea as a pest and take direct actions against them.

Historically, attempts to identify 'nuisance' individuals among flocks were made, and in a worst-case scenario, individuals were either relocated or removed. However, identification of unbanded individuals is unreliable and some kea have been subject to lethal removal, only for the problem to continue (Reid 2019). Currently there is no consistent approach to identifying or dealing with 'nuisance' individuals. The Kea Conservation Trust established a Conflicts Resolution Programme in 2013 and have a kea conflict report system whereby members of the public can report an issue and members of the Trust will work with the complainant towards transforming the situation. Some members of the public do not feel comfortable talking to DOC directly if they are having issues with kea and this is where KCT can be particularly valuable. Where DOC staff are notified directly of an incident, local staff often work with the landowner and KCT to try to transform the situation.

Some regions, such as northern and western Te Waipounamu, receive regular notifications of conflict with kea (c. 20 incidents per year), whilst others receive relatively few (c. 3–4 incidents per year). Essentially, human conflict with kea arises when their habitats overlap, such as at farms, forestry blocks, ski fields, alpine and resort villages, camping areas, and tourist stops.

Time of year may also be an important influence in when these incidents occur, with summer – early autumn being a particularly high-conflict time (Andrea Goodman, KCT, pers. comm.).

4.3.1 Farming

Kea have been long known to attack sheep in a phenomenon commonly referred to as 'kea strike'. Kea strike typically involves kea landing on the sheep's back between the shoulder and the pelvic region and can result in sheep illness, injury, and death in two ways; either by a) causing trauma and/or associated infection from wounds when pecking at the sheep's flesh in an attempt to reach fatty tissue, or b) by initiating or causing blood poisoning arising from infection with the soil dwelling bacteria *Clostridium* spp. (Grant 1993; Peat 1995). Kea strike occurs at a low and variable rate among high-country farms, though the risk factors are not well understood (Reid et al. 2020). Farming practices such as inoculating sheep against the bacteria *Clostridium,* removing carcasses from pastures promptly (so that kea do not get the opportunity to learn that sheep are food), not wintering stock at higher altitudes, and shearing during higher risk periods (so that kea cannot hold on to the sheep's fleece) are all actions that may mitigate risk. Trials using the bird repellent anthraquinone applied to the fleece of sheep have been conducted, though results were inconclusive (KCT, unpubl. data). Local kea density, distribution, and altitude may also play a role (Reid et al. 2020).

Kea getting into baleage is very common, resulting in high-conflict situations. This issue is not limited to high-country farms; it also occurs at lowland coastal sites.

Federated Farmers agreed to the granting of fully protected status to kea in 1986 on the condition that farmers who experience problems with kea strike on their stations may apply to DOC for permission to remove kea on a case-by-case basis (Miskelly 2014). A current policy or agreed approach on how to deal with 'nuisance' kea on farms is lacking, though DOC encourages the mindset of learning to live with kea and prefers to work with farmers to attempt to resolve any issues. Allegedly, a few farmers continue to poison and shoot kea illegally (Ray Molloy, DOC, pers. comm.).

4.3.2 Forestry

Kea who visit forestry operations are usually groups of explorative young birds that can cause conflict by interfering with equipment and distracting workers (Tamsin Orr-Walker, pers. obs.). In 2017, KCT, New Zealand Forest Owners Association (FOA), and forestry companies in the Nelson-Tasman Region published a set of guidelines for plantation forestry crews that outlines key protocols to be followed by forestry operations to avoid negative effects on kea and forestry equipment (KCT 2017). However, conflict situations with kea at forestry operations continue to arise, particularly in northern Te Waipounamu (Andrea Goodman, KCT, pers. comm.).

4.3.3 Ski fields

Conflict at ski fields arises when kea interfere with unattended vehicles, ski gear, equipment, and infrastructure. Prior to legal protection, kea were frequently shot by less tolerant ski field operators (Peat 1995). Increased understanding of the value and threatened status of kea, combined with a significant decline in numbers of kea observed at ski fields in recent years has led to a shift in perception amongst most ski field operators. NZ Ski has recently embarked on a 5-year project in partnership with KCT to improve conservation efforts for kea in the vicinity of The Remarkables and Coronet Peak ski fields, in Central Otago, and Mt Hutt, in Canterbury.

4.3.4 Other conflicts

Kea continue to be intentionally injured and shot by members of the public who have problems with the birds and take matters into their own hands. Massey University received 26 kea with fatal ballistic and blunt trauma injuries between 1990 and 2021 (Appendix: accessed

23/12/2021, search “kea, 1990 to present”, pathologists SA Hunter, MR Alley, BD Gartrell, F Castillo-Alcala, MG Collett, PHG Stockdale, RJ Norman, unpubl. necropsy reports). In 2011, five young kea were shot and dumped on a picnic table at Klondyke Corner near Arthur’s Pass. In 2017, a Tākaka Hill landowner was prosecuted for shooting at eight kea who were causing damage at his property, killing one. Originally, the landowner was feeding these kea to attract them to his property (Andrea Goodman, KCT, pers. comm.).

In 2016, KCT established a Conflicts Resolution Coordinator role, based in Nelson Tasman, to provide support to property owners and practical help on kea-proofing property and discouraging kea from hanging around. However, these incidents continue to occur in the region, and in November 2021, two kea were found shot dead in Golden Bay and handed in to DOC by a member of the public. Recently, a group of juvenile kea were observed playing with and consuming pink batts on a private lawn in *Ōkārito*, South Westland. The pink batts had been allegedly placed there by the property owner who was concerned that kea were damaging properties within the village (Kerry Weston, pers. obs.).

4.4 Living with kea – knowledge gaps

Based on the review above, the following are gaps in our knowledge that require further research and tool development to assist in recovery actions:

- Research ways to educate and increase public awareness of kea to reduce risk from human impacts and inform a collaborative advocacy strategy.
- Research on the variables potentially influencing kea strike and further mitigation approaches to this problem.
- A review of the tools and methods that have been trialled (either formally or informally) to deter kea and minimise property damage.
- Develop effective tools to minimise property damage using deterrent methods (e.g. repellents, coatings etc.).

5. Conclusions

The aim of this review was to summarise current knowledge of kea ecology and threats and to identify key areas that require further research to assist in recovery actions.

Whilst a solid knowledge base of research has been formed through past decades, substantial knowledge gaps remain. In part, this likely reflects the sparse distribution of kea over a range of habitats, their mobility, social structure, and variable conspicuousness. However, with improved technologies and the development of tools to record and share information, some of these barriers can be overcome. The key knowledge gaps identified within this review pertain to measuring population size and trends, movement ecology and habitat use, invasive predator ecology and control, lead impacts and sources, climate change impacts and human-kea interactions.

There is strong evidence that the primary threat to kea recovery is introduced predators, though the spatial and temporal elements of predator pressure and guild composition are still not well understood across the range of kea habitats. Several other key threats such as lead poisoning, human-kea interactions and climate change are also likely to limit kea recovery both directly, and via interactive effects. Vital to the recovery of kea will be understanding the relative impacts of these threats to kea across their range and measurement of long-term population responses to adaptive management.

The recent finding that adult kea are prone to predation by both stoats and feral cats in montane beech-forested valleys in eastern areas is an important implication. Understanding how predator suites differ among different kea habitats or among sub-populations, and when and where predators should be managed, is a priority. At present, predator control using aerial 1080 is the most effective tool for managing predator impacts across large areas of kea habitat, though the long-term benefits of 1080 for kea need to be measured within different habitats/sub-populations and carefully balanced against non-target risk. Understanding and managing the factors contributing to this risk should remain a priority for research and management.

Lead poisoning remains a key threat to kea, and lead from various sources remains widespread throughout kea habitat. The effects of lead exposure on kea reproductive capacity, survival and population growth are largely unstudied, however multiple studies document negative effects across these parameters in scavenging birds worldwide. Understanding the relative contribution and impacts of different lead sources and identifying problem areas is a priority for kea recovery. A nationally structured approach to lead-testing and survival monitoring would better inform management actions.

The most important implication of climate change for kea is the potential for increased predator pressure. Increases in temperature may lead to fewer periods of naturally lower rodent abundance within kea habitat, and subsequently, increases in invasive carnivores. Kea living within warmer, lower elevation forests are likely to be most affected, though a better understanding of population responses to these and other potential climate-related impacts are required.

Research continues to highlight the human influence on the ongoing decline of kea. Kea living near humans have increased opportunity to scrounge for human foods which increases the risk of poisoning, human-kea conflict, accidental injury and death. Improved management of human-kea interactions underpinned by behavioural and social science research; and encouraging a mindset of 'learning to live with kea' will be key to recovery.

It is recommended that the next step in the recovery of kea should be the development of a formal recovery strategy. However, to enable the full implementation of a kea recovery strategy, the identified knowledge gaps must be addressed and prioritised. Given the diverse array of subject matter and research disciplines involved, collaboration, partnership and tenacity will be key to success.

We recommend that a strategic, collaborative, and spatially structured research programme is developed and appropriately resourced, to enable a fuller understanding of kea throughout their range and measure population responses to management. Only then will we be able to recover this taonga.

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Appendix

DOC–Massey National Wildlife Pathology database information

DIAGNOSIS	CAPTIVE	WILD	TOTAL
Degenerative	**6**		**6**
degenerative	6		6
Iatrogenic	**2**		**2**
medical mishap	2		2
Infection	**23**	**13**	**36**
eye	1		1
infection	5	8	13
intestinal insult	1		1
liver	2		2
lung	1		1
renal	2		2
reproductive	1		1
respiratory	9	5	14
yolk sac infection	1		1
Metabolic	**1**	**6**	**7**
emaciation		4	4
starvation		2	2
stroke	1		1
Neoplastic	**7**		**7**
neoplasia	7		7
Parasitic	**2**		**2**
parasitic	2		2
Toxicity	**3**	**68**	**71**
1080		28	28
chocolate		2	2
lead	2	32	34
toxicity unknown	1	6	7
Trauma	**5**	**55**	**60**
ballistic		11	11
band injury	1	1	2
misadventure		4	4
trauma head	3	3	6
trauma intraspecific	1	1	2
trauma predator		5	5
trauma trap		3	3
trauma other/unknown		9	9
trauma vehicle		18	18
Open	**7**	**28**	**35**
open	7	28	35
TOTAL	54	170	224

Diagnostic categories: degenerative – the slow degrading of body structure and function due to age; iatrogenic – damage due to medical/human intervention during treatment of disease; infection – presence of, or damage caused by pathogens (germs); metabolic – disruption or decline of normal biochemical pathway in the body; neoplastic – presence of cancerous cells; parasitic – presence of parasites causing disease; toxic – the presence of toxin and/or tissue damage attributable to toxic effects; trauma – damage to tissues and structures caused by force; open – no clear cause of death could be determined from necropsy (source: Department of Conservation/Massey University).