4 Reintroduction of a threatened prey species to a predator-packed system: use of soft-release enclosures to minimise risks to establishment.

Authors & Affiliation

Katherine T. Bickerton[1,2,3], John G. Ewen[1], Stefano Canessa[4], Nik C. Cole[5,6], Jim J. Groombridge[3] & Rachel McCrea[7]

1. Institute of Zoology, Zoological Society of London, Regent's Park, London, NW1 4RY, United Kingdom
2. School of Mathematics, Statistics and Actuarial Science, University of Kent, Canterbury, Kent, CT2 7FS, United Kingdom
3. Durrell Institute of Conservation and Ecology, School of Anthropology and Conservation, University of Kent, Canterbury, Kent, United Kingdom
4. Division of Conservation Biology, Institute of Ecology and Evolution, University of Bern, 3012 Bern, Switzerland
5. Durrell Wildlife Conservation Trust, Les Augrès Manor, Trinity, Jersey, United Kingdom
6. Mauritian Wildlife Foundation, Grannum Road, Vacoas, Mauritius
7. Department of Mathematics and Statistics, Lancaster University, Lancaster, LA1 4YR, United Kingdom

4.1 Abstract

Conservation translocations are increasingly used as a conservation tool, however reptiles, especially prey species, are underrepresented in published literature. In addition, many reptilian prey species are small and highly cryptic, especially squamates, leading to difficulties with post-release monitoring and conservation management. In this chapter, we describe the planning process, translocation and initial monitoring for a reintroduction of an endemic threatened lizard species, the lesser night gecko (*Nactus coindemirensis*), to Round Island, Mauritius. Our release strategy was informed by interviews with herpetofauna experts and aided in highlighting key risks to establishment, especially predation and dispersal. Soft-release enclosures with predator proof fencing and intensive predator removal prior to release were used to minimise these risks and post-release monitoring was carried out using capture-recapture surveys. We use geometric removal models to estimate the proportion of predators removed and Cormack-Jolly-Seber models to monitor survival in the seven months following release. We calculated minimum convex polygons to measure dispersal of the populations in each enclosure. We estimate we removed 67-75% of the most numerous predator in each enclosure. Estimates of monthly survival probability varied between 0.747-0.897 for females and 0.792-0.918 for males, dependent on release enclosure. We had difficulties estimating dispersal due to few individuals being recaptured within surveys. Our initial survival estimates, in addition to the presence of gravid females and juveniles, indicate we have, at least partially, limited predation and dispersal that may have otherwise prevented early persistence. We depend on continued monitoring to further understand the dynamics of the population to inform future management and the next stages of the release, and emphasize the importance of having a long-term post-release monitoring plan in place to allow for accurate measures of success.

Key words: reintroduction; conservation translocation; soft-release; predation; dispersal; removal; Mauritius; lesser night gecko; *Nactus coindemirensis*

4.2 Introduction

Conservation translocations are defined as the intentional movement of organisms from one location for release in another with the primary objective of improving the conservation status of the species and/or restoring ecosystem function (Seddon 2010; IUCN/SSC, 2013). Translocations are increasingly used as a valuable tool in the conservation of threatened species (Seddon *et al.* 2007; Seddon 2010; Taylor *et al.* 2017). Reintroductions are defined as translocations where species are released within their historic range with the aim to restore extirpated populations (Armstrong & Seddon 2008). During the last two decades, publications on conservation translocations, in particular reintroductions, have dramatically increased (Seddon *et al.* 2012) and may even underrepresent the true number of translocations taking place due to practitioners not publishing their projects and to publication bias towards successful actions (Fischer & Lindenmayer 2000; Bajomi *et al.* 2010). Additionally, there have been taxonomic biases in the species translocated (Seddon *et al.* 2005), with mammals and birds accounting for 93% of published animal translocation studies prior to 2000 (Fischer & Lindenmayer 2000) and continuing to be overrepresented in the literature (Bajomi *et al.* 2010; Bubac *et al.* 2019; Evans *et al.* 2023).

Reptiles are one of the groups underrepresented in the translocation literature (Bubac *et al.* 2019; Evans *et al.* 2023) despite approximately 21% of reptile species being threatened (IUCN, 2022). Historically, translocations of reptiles have had very low or unknown success rates (Germano & Bishop 2009) and efforts have largely been focused on the most threatened orders: Testudines and Crocodylia (Cox *et al.* 2022); 50% of case studies collated by the IUCN SSC Conservation Translocation Specialist Group focused on translocations of species from these groups (Soorae 2008, 2010, 2011, 2013, 2016, 2018, 2021). Reintroductions of Squamata, the order including lizards and snakes, have been undertaken but have rarely been published, which may be partly due to the presence of fewer charismatic species within this order, leading to publication bias (Seddon *et al.* 2005) and many species exhibiting highly cryptic behaviour (Bilby & Moseby 2022) making effective post release monitoring more challenging. In particular, there is very limited information in the literature on translocations of species in the family Gekkonidae, with the majority of translocations being undertaken in New Zealand for long lived species (Knox & Monks 2014; Flynn-Plummer & Monks 2021). Published reports for this taxon document the difficulties in tracking species due to their small size and cryptic behaviours (Towns *et al.* 2016).

A key challenge with reptile reintroductions, especially when considering smaller cryptic species, is that many reptiles are omnivorous and relatively low in the food chain, meaning predation pressure must be considered. Predation is known to suppress prey populations, especially following severe declines when population abundance is low (Sinclair *et al.* 1998). Therefore, establishing new populations of prey species from small numbers of founders, as is often the case in reintroductions of threatened species (Deredec & Courchamp 2007), is less likely to succeed when predators are present (Tobajas *et al.* 2020). The presence of predators can increase stress and lead to post-release dispersal away from release sites or ranging more widely (Dickens *et al.* 2010), causing fragmentation of the release population, which is particularly concerning in squamate reptiles as those species with the highest risk of extinction also tend to have the smallest home ranges (Bohm *et al*. 2016). This scenario can lead to a lack of individuals with overlapping home ranges which in turn can lower levels of reproduction and the ability of the population to establish (Bilby & Moseby 2023). In several cases, suppression or removal of predators, most often carried out when predators were non-native, has increased the chance of establishment (Sinclair *et al.* 1998; Tobajas *et al.* 2020). However, this approach may not be possible in systems where predators are native and threatened. Furthermore, the majority of reintroduction literature only reports the outcome of the translocation population, with no reference to ecosystem level impacts (Taylor *et al.* 2017). This is particularly important in systems where species that directly interact with the focal study species are also threatened, as negative interactions could have broad implications for the release ecosystem (Amstrong & Seddon 2008; Taylor *et al.* 2017).

Post-release monitoring can also be difficult to achieve for squamate reptiles, especially when higher rates of post-release movement are coupled with cryptic behaviours and small body size. Post-release monitoring is important because it allows for demographic parameters, such as survival, population size and recruitment, to be estimated, which gives managers the ability to determine the best possible actions to conserve the population. Additionally, monitoring of individuals' health following release allows managers to choose actions that maximize animal welfare. Monitoring and estimation of demographic parameters and health frequently rely on individuals being uniquely identifiable, especially for accurate measures of survival and population size (Pollock *et al.* 1990; Schwarz & Arnason 1996). Some reptile species have unique markings, allowing for identification (Bolger *et al.* 2012; Urian *et al.* 2015). For those that do not have these markings, a variety of internal and external tagging can be employed including PIT tags, bead tagging, toe clipping, elastomer implants, etc. (Ferner & Plummer 2016). However, for smaller reptile species, many internal tags are too large or prohibitively expensive and for prey species, external markings may make them more visible to predators and therefore less likely to establish following a reintroduction. In most cases, monitoring of smaller species requires capture which can be very labour intensive.

A variety of release strategies can be applied when translocating animals, aimed to best support initial establishment and ultimately improve success (Moseby *et al.* 2014). The terms *hard-release* and *soft-release* indicate, respectively, releasing animals straight into the release site versus holding the animals at the release site under controlled conditions (e.g., in pens or aviaries) to allow for acclimatisation (Griffith *et al.* 1989; Armstrong & Seddon 2008). In addition to this, habitat modifications, artificial refugia and supplementary feeding have been implemented in many cases across many taxa (Armstrong & Seddon 2008; Moseby *et al.* 2014). Soft-release approaches have been used to minimise both predation and dispersal when translocating prey species with mixed results (de Milliano *et al.* 2016; Bilby & Moseby 2023), alongside habitat modification. There is evidence from several translocations of one lizard species (*Naultinus gemmeus*; Knox & Monks 2014), and one tortoise species (*Gopherus polyphemus*; Tuberville *et al.* 2005), that using a soft-release pen can help to anchor release animals to the site and reduce dispersal when compared to hard-release of individuals (Knox & Monks 2014; Knox *et al.* 2017; Nafus *et al.* 2017; Flynn-Plummer & Monks 2021; Resende *et al.* 2021; Linhoff & Donnelly 2022). Importantly, there are very few additional studies that provide comparisons between strategies, because the species being translocated are usually threatened, so the risk of using methods which may not be successful may be perceived as too high despite the potential gain in knowledge. This scenario therefore leads to high uncertainty when planning a squamate translocation, meaning that scarce available literature usually must be integrated with expert opinion (Runge *et al.* 2011).

In this chapter, we assessed the immediate post-release outcome of using soft-release enclosures to support releases of a small, threatened prey species endemic to Mauritius, the lesser night gecko *Nactus coindemirensis*. We translocated 120 geckos from a wild population on the island of Gunner's Quoin to four soft-release enclosures on Round Island. Our aim was to minimise predation and dispersal of the release population and use the lesser night gecko as a model system for the reintroduction of a prey species into a system with native predators, using expert surveys to inform decision making where empirical data was not available. Following the IUCN conservation translocation guidelines (IUCN/SSC 2013), the translocation process entailed removal of threatened endemic predators and post-release monitoring. Our study reports initial analyses of survival and dispersal in the seven months following release.

4.3 Methods

4.3.1 Study species, source & destination sites

Lesser night geckos (*Nactus coindemirensis*) are the smallest of the three *Nactus* species endemic to Mauritius and currently classified as Vulnerable on the IUCN red list (Cole *et al.* 2021). Lesser night geckos are a cryptic, nocturnal gecko species with a snout-to-vent length of approximately 29 mm and uniquely identifiable dorsal patterns (Bullock *et al.* 1985). The species was believed to be present across Mauritius prior to human colonisation and before the introduction of invasive predators, such as rats (*Rattus rattus* and *R. norvegicus*), cats (*Felis domesticus*) and mongoose (*Herpestes javanicus*), and competitors, such as the Asian house gecko (*Hemidactylus frenatus*; Bullock *et al.* 1985; Cole *et al.* 2005). There are currently three remnant populations of lesser night gecko, two on the northern islands of Gunner's Quoin and Pigeon House Rock, and one on the southeastern islet of Ilot Vacoas, in addition to a reintroduced population on Île Marianne (Figure 4.1; Cole *et al.* 2021).

The Gunner's Quoin population is the largest and most stable population (following the eradication, in 1995, of rats which had suppressed the population; Cole *et al.* 2005), consisting of approximately 12000-14000 individuals (Cole *et al.* 2007). The Île Marianne population was the result of a combined translocation of wild individuals from Îlot Vacoas and captive reared individuals from Jersey Zoo, in April 2011 (Bickerton *et al.* 2023). Both the southeastern islets are small with limited resources and population size estimates of approximately 400-600, which are likely approaching carrying capacity. Despite all of the islands being closed to the public, they are easily accessible from the main island so regularly trespassed (with the exception of Pigeon House Rock, which is difficult to access and therefore not regularly monitored), increasing the risk of incursions by invasive species. Lesser night geckos for the translocation were sourced from Gunner's Quoin, due to its stability and proximity to Round Island.

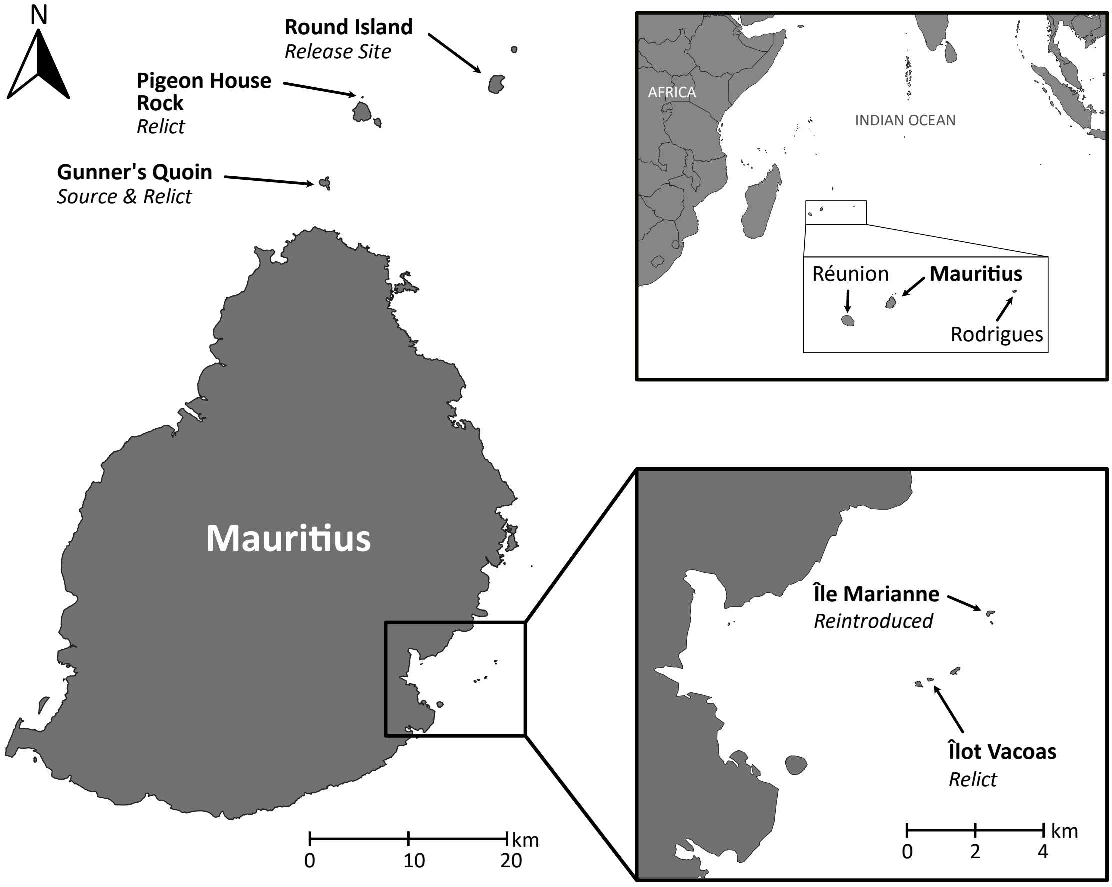

Figure 4.1: Map of Mauritius and outlying islands on which relict or reintroduced populations of lesser night gecko *Nactus coindemirensis* are present. The source island for the reintroduction, Gunner's Quoin, and release island, Round Island, are indicated.

Round Island is the only closed island nature reserve in Mauritius with a constant presence of wardens, which prevents trespassing and increases the chance of invasive species being detected and prevented from spreading. Although lesser night geckos are not present on Round island, it is believed they would have been previously as suitable habitat is present and their sister species, *Nactus durrellorum*, is present, which fills a similar niche but is of larger size. Lesser night geckos are thought to have gone extinct from Round island due to the presence of goats and rabbits which reduced vegetation coverage (Bullock *et al.* 2002), destroying key refugia leading to higher predation. Round island is also home to the only near intact reptile community in Mauritius (Bullock 1986). Restoration of the island began in the 1970s, greatly improving vegetation coverage and allowing populations of reptiles present to increase. Restoration is now guided by the Round Island Management Plan (Round Island Management Plan 2019-2025), a 100-year plan, split into 5-year subsections, which encompasses all aspects of restoration and includes restoring the wider reptile community, especially prey species such as the lesser night gecko.

4.3.2 Translocation method & site selection

All field and social science methods were ethically approved by the University of Kent's School of Anthropology and Conservation Research Ethics Committee (#20221664877579172, approved 4th October 2022). To obtain an initial assessment of the most optimal reintroduction strategy for lesser night gecko on Round Island, we carried out a semi-structured interview with herpetofauna experts at the 9th World Congress for Herpetology 2020 in New Zealand. As a large proportion of conservation translocations of reptiles have been carried out in New Zealand and the world congress was taking place in New Zealand, in addition be being the largest meeting for herpetologists globally, we selected experts from those attending the conference. Experts were defined as those with knowledge of threatened reptile translocations from practical experience. They were selected from the conference attendees using three criteria: those who had published on reptile translocations; those who were presenting research on reptile translocations; and those who attended the symposium on translocations and volunteered their expertise following a call for interviewees during the symposium. Interviewees from the first two criteria were contacted prior to the conference by email to ask if they would be willing to contribute. All respondents had experience in translocation of threatened species, with 24 working on reptiles. Just over half of the experts (n=14) were professional ecologists working as consultants or scientific advisors in public and private sectors. The remaining 12 held academic positions at universities or research institutes. A total of 26 experts were interviewed, from 20 different organisations across Europe, North America and Oceania.

All interviews were conducted individually in person at the conference (with the exception of one that was unable to attend and was carried out via Skype after the conference). Interviewees were provided with an information pack about the study species and source and destination sites (4.6.1 Supplementary Information: Figure S4.1). The questions were formulated from our short-term objective for the translocation: to minimise the loss of translocated individuals to predation at the release site. Following the results of the survey, minimising dispersal also became a key part of our objectives. Our interview consisted of two key parts. Firstly, what method would experts use to undertake the translocation; secondly, what risks they thought would be associated with these methods (4.6.1 Supplementary Information: Figure S4.2). Following evaluation of the results of the interviews, we decided to use soft-release enclosures with predator proof fencing and artificial refugia to enhance the available habitat. We had planned to carry out a more formal expert elicitation process using the Delphi method (Burgman *et al.* 2011), to elicit quantitative estimates of survival under different release scenarios following these interviews however this was not possible due to the uncertainty surrounding whether the translocation would occur (due to the Covid-19 pandemic) during the time constraints of the project.

Four release enclosures (each 10x15 m) were selected based on habitat suitability and ability to construct predator proof fence (Table 4.1; Figure 4.2). The fences for each enclosure were constructed from galvanised 1-m steel posts, placed approximately 1 m apart, and galvanised steel mesh. The outside of the fences was covered with heavy duty black plastic from the ground (where 30 cm were buried under soil) to 70 cm high. The buried parts were weighted with rocks to prevent predators from burrowing under the plastic and climbing the inside of the fence. The plastic was used to deter Telfair's and Bojer's skinks (*Leiolopisma telfairii* and *Gongylomorphus bojerii*) and keel-scaled boas (*Casarea dussumieri*), as these predators struggle to climb slippery surfaces. On the inside of the fence, plastic was used to cover the bottom 30 cm of the fence and again buried at the base, to prevent the lesser night geckos from climbing out. The top 10 cm of both sides of the fences were coated in lithium grease to prevent geckos from climbing over, as their toe pads are unable to adhere to greased surfaces. Artificial refugia were created by coating large cardboard egg trays in cement, stacking 7-8 trays at slight angles to create multiple microclimates within each stack, then weighting them with a rock. These type of artificial refugia had previously been used with lesser night geckos in captivity and were easy to transport across the island. We used these to bridge gaps between patches of high-quality habitat in each enclosure, using 5-10 stacks per enclosure dependent on enclosure size (4.6.2 Supplementary Information: Figure S4.3). Lesser night geckos are rarely observed more than 1m from suitable refugia (Cole 2005) therefore it was important to provide refugia in areas of open habitat where natural refugia was limited. The selected sites predominantly contained continuous habitat however in the dry season, a lack of vegetation led to gaps between suitable habitat, in which we used the artificial refugia as bridges.

Figure 4.2: Locations of lesser night gecko *Nactus coindemirensis* release enclosures and field station on Round Island, Mauritius. Fenced areas shown on the right, square points on the outline indicate fence posts, spaced at approximately 1 m, and dotted lines indicate the grid system used to monitor animal movements (grid markers were placed along fences only to prevent disturbance to habitat within the enclosures).

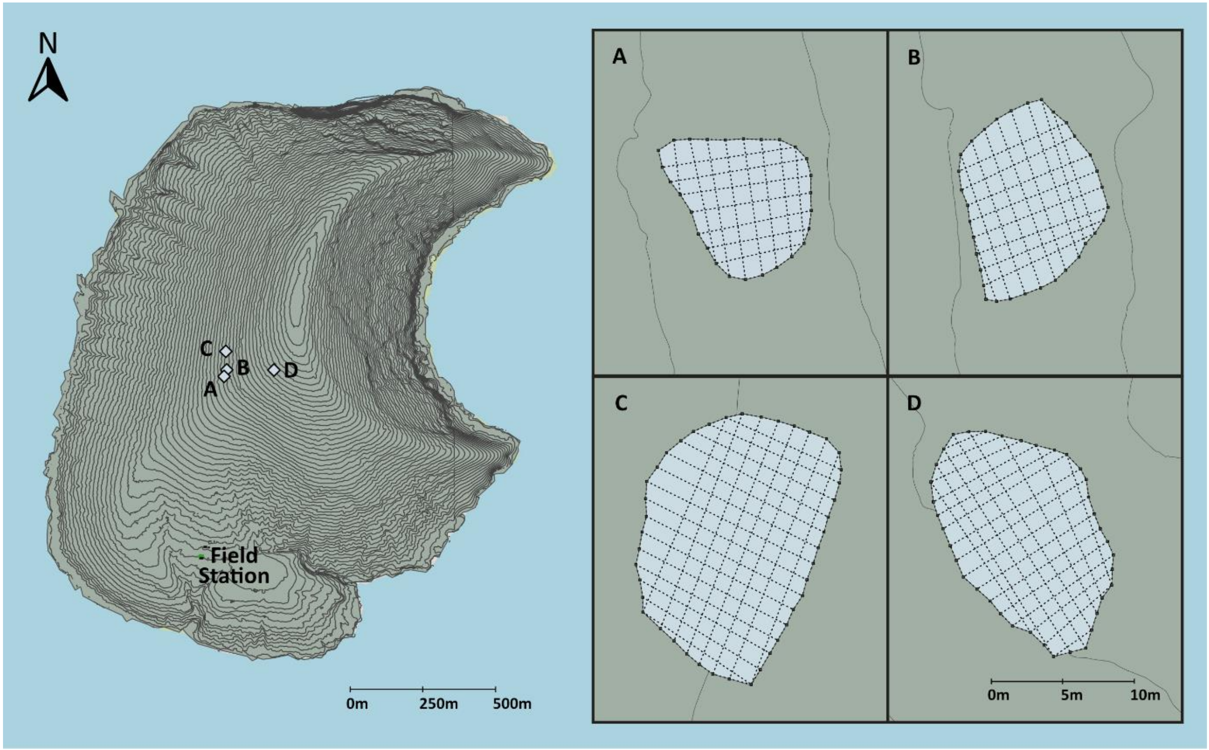

Table 4.1: Selection criteria for lesser night gecko release enclosures on Round Island.

Criteria	**Description**
Refugia	Rocky outcrops with crevices of between 1-2cm in width and at least 30cm in depth (Chapter 5, Cole 2005), long grasses and leaf litter. As vegetation cover varies seasonally, areas were selected in December at the end of the dry season when vegetation cover is very low, and checked again for suitability in April, near the end of the wet season when vegetation cover is high.
Food availability	Lesser night geckos are insectivorous and have a similar diet to the Durrell's night gecko, taking smaller instars of the same prey species (Chapter 5, Cole 2005). Little work has been carried out on the insects of Round Island therefore we used the presence of Durrell's night gecko with good body condition as a proxy for food availability. Individuals were captured in December 2022 and April 2023, weighed and measured and their body conditions compared to average body conditions of the population.
Low predator density	Selection of areas of the island where lower densities of predators had been recorded, the main predators being the keel-scaled boa *Casarea dussumieri*, Telfair's skink *Leiolopisma telfairii* and Gunther's gecko *Phelsuma guentheri*. We avoided the palm rich forests where the highest predator densities were recorded and focused on the rock-slab and summit areas where predator densities were lowest (Figure 4.2; Cole *et al.* 2018c). Juveniles of these predators also of high risk to lesser night geckos so we aimed to avoid areas where they may lay their eggs. Gunther's geckos are mainly arboreal, so minimising palms within sites was important and high rock ledges close to palms (Carpenter *et al.* 2003). Telfairs skinks and Keel-Scaled boas are more likely to use soil which is particularly scarce in both slab and summit habitats. Bojer's skinks *Gongylomorphus bojerii* are widespread across the island but are less likely to predate lesser night geckos as they are diurnal.
Connectivity	Although there is high predator density in the palm rich areas of Round Island (Figure 4.2), the highest density of Durrell's night geckos are present here (Cole *et al.* 2018c) and likely the highest density of prey therefore we looked for sites close to palm rich areas.
Level terrain	The terrain on Round Island is very uneven with large steps in the rock therefore we aimed to find areas with suitable rock outcrops and a perimeter with steps of less than 30cm, so that we could prevent gaps in the fence that predators could enter through.
Water run-off	In the wet season, heavy rains can lead to run-off in specific areas of the island, potentially washing away loose refugia. The surveys in April enabled us to identify these areas and avoid them where possible.
Tortoise activity	Aldabra giant tortoises have been present on Round Island since their introduction in 2000 (Griffiths *et al.* 2010) and are now widespread across all habitats. Other experimental fenced areas have been damaged by tortoises, so we aim to avoid areas of high activity (such as the artificial wallows on the island, Figure 4.2) and use an opaque covering on the fences so any potential food items within the enclosures are not visible.
Seabird activity	Round Island is an important breeding site for several species of threatened seabirds, therefore we avoided breeding areas to prevent disturbance.
Proximity to field station	Due to the difficulty in accessing parts of the island, especially in the wet season, we chose sites on the south side of the island to allow for easier access during the build, release and post-release monitoring.

After construction, any predators found within the enclosures were removed by hand. We aimed to remove all Telfair's skinks, Gunther's geckos (*Phelsuma guentheri*) and keel-scaled boas, and to reduce the Bojer's skink population. Bojer's skinks are smaller and more difficult to catch, therefore complete removal would not have been possible without compromising refugia for the lesser night geckos. Removal of Bojer's skinks was carried out by 1-3 experienced surveyors with the aim to capture and remove as many as possible with minimal damage to the release enclosures. Surveys were carried out during peak activity times for Bojer's skinks (8:00-12:00 and 15:00-18:00; Cole & Payne 2022) for approximately 45 minutes, with a minimum of a 15-minute break between surveys. Each individual captured was recorded, photographed for later recognition (Bojer's skinks have unique dorsal patterns), and sexed, then released within 10 m of the enclosure. Although abundant on Round Island, Bojer's skinks are classified as Critically Endangered on the IUCN Red List (Cole & Payne 2022); and because they exhibit territorial behaviour, releasing them further than 10 m from the enclosure may have led to increased competition and reduced body condition. All enclosures were surveyed as many times as feasible (3-9 surveys) in the two weeks before the translocation, to ensure as many individual captures as possible (4.6.3 Supplementary Information: Table S4.1). There were some additional removals outside of the targeted surveys, which will be referred to as incidental removals. Following the surveys, the photo identification software "hotspotter" (Crall *et al.* 2013) was used to check for multiple captures of the same individual which would indicate the fences were not secure. Removal of Telfair's skinks, Gunther's geckos and keel-scaled boas occurred whenever individuals were encountered due to their variety of life histories (including if encountered during Bojer's skink removal surveys), and their distinctive markings were recorded (4.6.3 Supplementary Information: Table S4.2). There were no recaptures following removal therefore we were not able to assess if there had been any impact on predator body condition following removal.

4.3.3 Translocation procedure

Geckos were collected from across Gunner's Quoin to maximise genetic variation in our release population. Healthy adults, ideally with intact tails and toes, were selected, as the translocation process would likely be too stressful for juveniles. Gravid females were only selected if they were in the early stages of gravidity, as those in later stages were likely to be more easily stressed. Each individual captured was processed (sexed, weighed, snout-to-vent length and tail length measured, and photographed) then placed in individual, labelled cloth reptile bags for transportation. Geckos were translocated by helicopter or boat either the morning or two mornings after capture (Table 4.2). On arrival, the plastic box containing the individually bagged geckos was taken directly to the quarantine facility and checked for biological material. Bags were not opened prior to release to minimise stress. The geckos remained in the reptile bags in a shaded area of the quarantine room until the release in the evening. All captures and translocations took place between 30 October and 4 November 2022 (Table 4.2).

Table 4.2**:** Number of lesser night geckos captured on Gunner's Quoin and translocated to Round Island, captures all take place between 19:00 and 02:00 on the date of capture and release starting at 19:30 on the date of release. Individuals released in different enclosures were transported in separate containers.

<table>
<tr><th colspan="2">Date</th><th colspan="3">No. Individuals</th><th rowspan="2">Transport Method</th><th rowspan="2">Enclosure</th></tr>
<tr><th>Capture</th><th>Transport & Release</th><th>Male</th><th>Female</th><th>Total</th></tr>
<tr><td>30/10/2023</td><td>31/10/2023</td><td>15</td><td>15</td><td>30</td><td>Helicopter</td><td>A</td></tr>
<tr><td rowspan="2">01/11/2023</td><td rowspan="2">02/11/2023</td><td>15</td><td>15</td><td>30</td><td rowspan="2">Boat</td><td>B</td></tr>
<tr><td>3</td><td>0</td><td>3</td><td>D</td></tr>
<tr><td rowspan="2">02/11/2023</td><td rowspan="3">04/11/2023</td><td>3</td><td>5</td><td>8</td><td rowspan="3">Helicopter</td><td>C</td></tr>
<tr><td>11</td><td>16</td><td>27</td><td>D</td></tr>
<tr><td>03/11/2023</td><td>11</td><td>11</td><td>22</td><td>C</td></tr>
</table>

Releases were carried out on three nights between 31 October and 4 November (Table 4.2). Geckos were released in male-female pairs at preselected locations, equally spaced within the most suitable habitat in each soft-release enclosure (Table 4.1). We aimed to release 30 geckos per enclosure to replicate the density found in the wild, and to not heavily impact the source population, with our harvest being <1% of the total population. Before release, each pair were removed from their bags and placed in a container lined with damp paper towel for 5 minutes to allow them to rehydrate, then released from the container with minimal contact to prevent stress. Releases started once completely dark, at approximately 19:00. A team of three field staff carried out each release with two people inside the enclosure and one outside ensuring predators did not enter via the access ladders. Location of release for each individual was recorded in addition to time of release and environmental conditions. To allow for accurate monitoring of release location and dispersal, each enclosure was divided into approximately 1-m^2 grid squares with a unique alphanumeric code. GPS coordinates were taken for the corners of each enclosure and averaged for accuracy (as the GPS units available were only accurate to 3 m), then coordinates of the centre of each grid square were calculated.

4.3.4 Post-release monitoring

To minimise stress and allow the released animals to acclimatise, enclosures were left undisturbed the night following release, then monitored externally for four subsequent nights. Monitoring consisted of looking for individuals within enclosures, recording locations and photographing where possible for identification. Predator checks were also carried out and enclosures entered only if a Telfair's skink or keel-scaled boa was spotted. After 5 nights, each enclosure was surveyed once internally, geckos seen were recorded and photographed but not captured to prevent stress, and any predators detected were removed. The enclosures were left undisturbed for two weeks to allow the geckos to habituate without the additional stress of surveys. During this period, predator checks were carried out, but an effort was made to minimize disturbance.

Mark-recapture surveys were carried out within the enclosures, with four surveys per enclosure over a survey trip of 2-3 weeks (weather dependent as enclosures are unsafe to access during and post heavy rains). Five survey trips were carried out between November 2022 and May 2023, with 3-4 weeks between trips (4.6.3 Supplementary Information: Table S4.3). All surveys were carried out by 1-2 members of staff, starting at least 30 minutes after dusk with the same survey route used around each enclosure (4.6.3 Supplementary Information: Figure S4.4), walked once to prevent recaptures of the same individual. The first 3 surveys were carried out by experienced surveyors and the 3rd survey was used to train a new surveyor who carried out the 4th and 5th surveys. Captures were attempted for all geckos sighted and upon capture, individuals were sexed, snout-to-vent length and tail length measured, weighed, checked for injuries, and photographed for identification, then released back to the sight of capture. Location of capture, using the grid system, was recorded and environmental variables measured (air temperature, substrate temperature, moon phase and visibility, cloud cover, wind and substrate type; 4.6.4 Supplementary Information: Table S4.5). Following surveys, we used Hotspotter was used to identify individuals from their unique dorsal pattern. As with earlier surveys, predators found within enclosures were removed and point of entry identified and sealed (see 4.6.2 Supplementary Information for full survey protocol).

4.3.5 Data analyses: removal

To assess the efficacy of our removal surveys, we modelled the Bojer's skink removals between 24 October and 4 November 2023 using a closed geometric removal model which accounted for variation in detection over time (Matechou *et al.* 2016). We used survey effort as a covariate, calculated as a scaled product of the number of surveyors and length of survey and obtained 95% confidence intervals for all parameters using a non-parametric bootstrap procedure (DiCiccio & Efron 1996). We chose to use a closed population model as none of the skinks were recaptured, therefore we assumed that once removed, they were unlikely to re-enter the enclosures. Incidental removals were not included as 85% of removals occurred during targeted surveys. We calculated the proportion of each population removed using the estimated total population and the number of individuals removed. The removals for each enclosure were modelled separately, as opposed to using an integrated approach (Zhou *et al.* 2019), as survey length, time of day, effort and frequency varied (4.6.3 Supplementary Information Table S4.1).

4.3.6 Data analyses: survival

To monitor population demographics, we used Cormack-Jolly-Seber (CJS) models (Cormack 1964; Jolly 1965; Seber 1965) to estimate survival ϕ and detection probability p. Although the population is open as juveniles were recorded, the use of CJS models is justified because at the time of our surveys there would have been no entries into the adult population. This allows us to infer the population size without the use of the Jolly-Seber model which requires modification for use with translocated populations (Bickerton *et al.* 2023). Capture histories were constructed by grouping all captures for each survey trip as individuals were rarely caught multiple times during the same trip, 1 was used to denote a capture and 0 to denote no capture. We assessed variation in ϕ and p with sex and release enclosure as group effects, and time as survey occasion. For ϕ we also considered the impact of cyclone Freddy which passed by Mauritius on 20 February 2023, between the third and fourth capture-recapture survey. This was modelled as a binary effect with the environmental state being 1 between the February and March surveys, and 0 for the remaining surveys. We assessed variation in p over time with survey effort, defined as the number of surveys per trip, surveyor, averaged environmental variables recorded during the surveys (air temperature, substrate temperature, moon phase, moon visibility, cloud cover and a qualitative measure of wind) and averaged environmental variables recorded by the weather station on the island (air temperature, wind speed, humidity and rainfall). Full covariate details in 4.6.4 Supplementary Information Table S4.5. All covariates were modelled using a *logit* link and rescaling estimates to account for uneven time intervals between survey trips. We build a model for each possible combination of covariates, and ranked models using Akaike's Information Criterion corrected for small sample size (AICc). We checked suitability of the highest ranked CJS model with our data using the 2.ct goodness of fit test from the R package

“R2ucare” (Gimenez *et al.* 2018), which was non-significant ($\chi^2 = 7.420, p = 0.060$) indicating the model is suitable. Test 3.SR could not be computed due to all individuals being included at time 1.

4.3.7 Data analyses: dispersal

To monitor dispersal of populations within each enclosure, we used an approximately 1 m^2 grid system within each enclosure (Figure 4.2). The grid square reference of release and each subsequent capture or sighting of known individuals was recorded and the straight-line distances between the centre of each grid square calculated. For each individual, cumulative distance moved was calculated, assuming movements were between the centre of each grid square. Due to low sample size, we were unable to analyse this data statistically. To monitor temporal variation in the extent of population distribution, we calculated minimum convex polygons (MCPs; Seaman *et al.* 1999) for each survey occasion for each enclosure. As number of captures was lower in the final two surveys, MCPs were scaled by the number of captures per survey. A general linear model was used to investigate the relationship between release enclosure area and total distance moved.

4.4 Results

4.4.1 Expert recommendations

Half of the 26 experts interviewed recommended using a soft-release enclosure and a further 27% recommended an experimental approach where soft- and hard-release methods were compared (Figure 4.3), as their first choice of release strategy. Soft-release enclosures were suggested, because they provide physical barriers to predators and can also limit post-release dispersal which may prevent reproduction given the lesser night geckos' small home range. A variety of enclosure types were suggested, including solid fences designed to prevent movement to and from enclosures, and "leaky" fences where small holes allow translocated animals to disperse from enclosures but prevent larger predators from entering. We chose to combine these suggestions, installing a solid fence for the initial post-release period, then planning to make the fence leaky once juveniles were sighted and dispersal had reduced. The experimental approach recommended by 27% of interviewees was intended to maximise the knowledge gained from the translocation. We chose not to use this approach due to the high risk of predation of the hard-released population, especially as lesser night geckos are threatened and the number we were permitted to move was limited (which would have also limited our ability to infer differences between methods). Artificial refugia were chosen by two experts as their first choice of release strategy, and by five experts as their second or third choice. The types of refugia suggested by the experts varied included building rock piles, filling natural crevices with small rocks to prevent predation, stacking roofing tiles and other artificial materials with holes drilled to small sizes to minimise predation, and placing the refuge within the range of the source population, then transferring the whole refuge to reduce stress. The later suggestion was not possible due to biosecurity concerns; however, all remaining suggestions were taken into account when designing our artificial refugia.

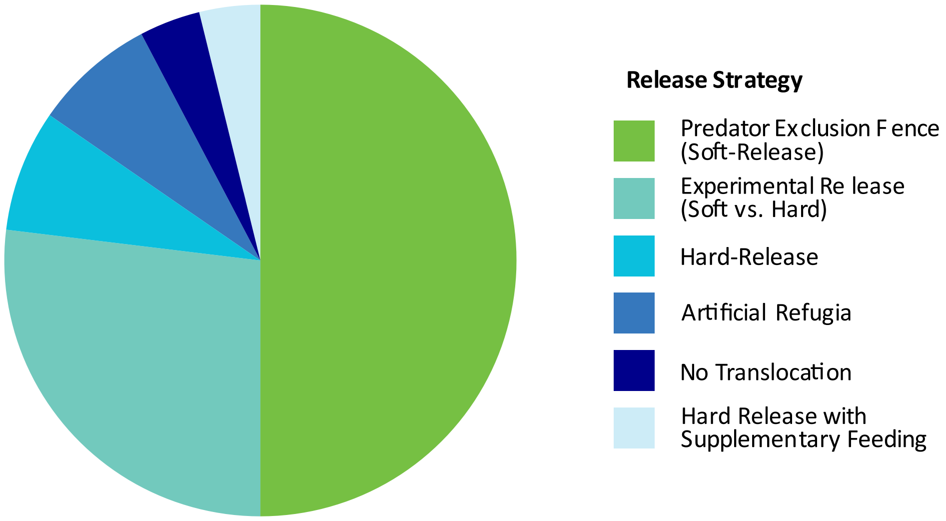

Figure 4.3: Recommended release strategy for the lesser night gecko translocation indicated as first choice by each of the 26 experts interviewed at the 9th World Congress of Herpetology.

The second part of our interview addressed the risks associated with the recommended release strategy and more broadly relating to the translocation. Of the 24 risks identified, the most common were dispersal (directly following the release due to stress), post-release monitoring (specifically relating to low detection probability of the species and the availability of funding for long-term monitoring), predation and disease (risks to both the translocated population and the reptile species present at the release enclosures). Measures were taken during the release to mitigate these risks, with our choice of release strategy aiming to minimise dispersal and predation. Post-release monitoring was carried out by experienced surveyors and timed for optimal detection. To prevent transfer of disease or parasites, only healthy geckos with high body condition scores were selected. Healthy geckos were defined as those with their original tail or no recent tail breaks, intact toes, no scarring and a mass to length ratio of above 0.02g/mm. All geckos were stored in individual cloth bags for one night before release and any excrement or organic matter found within them was disposed of safely.

4.4.2 Removal

Across the four release enclosures, we removed 220 Bojer's skinks, 26 Telfair's skinks and 2 Gunther's geckos; no keel-scaled boas were located. Our models estimated that we removed over 67% of the Bojer's skink population in each enclosure (Figure 4.4) and no individuals were recaptured, providing confidence that our enclosure fences were secure. The highest number of Bojer's skinks (99) were captured in our largest enclosure (C) where our removal model gave an estimated 75% reduction in population. Enclosure D was only surveyed three times due to time constraints however the reduction in capture rate between the 2nd and 3rd survey was large allowing for confidence that the majority of Bojer's skinks had been removed. Although we did not capture enough Telfair's skinks to run a statistical analysis, they are much larger in size and have a higher capture probability, therefore we are confident a minimum of 75% of Telfair's skinks would have also been removed. The survey effort covariate was positively correlated with captures in all models, with increased length of survey and number of surveyors increasing chances of capture.

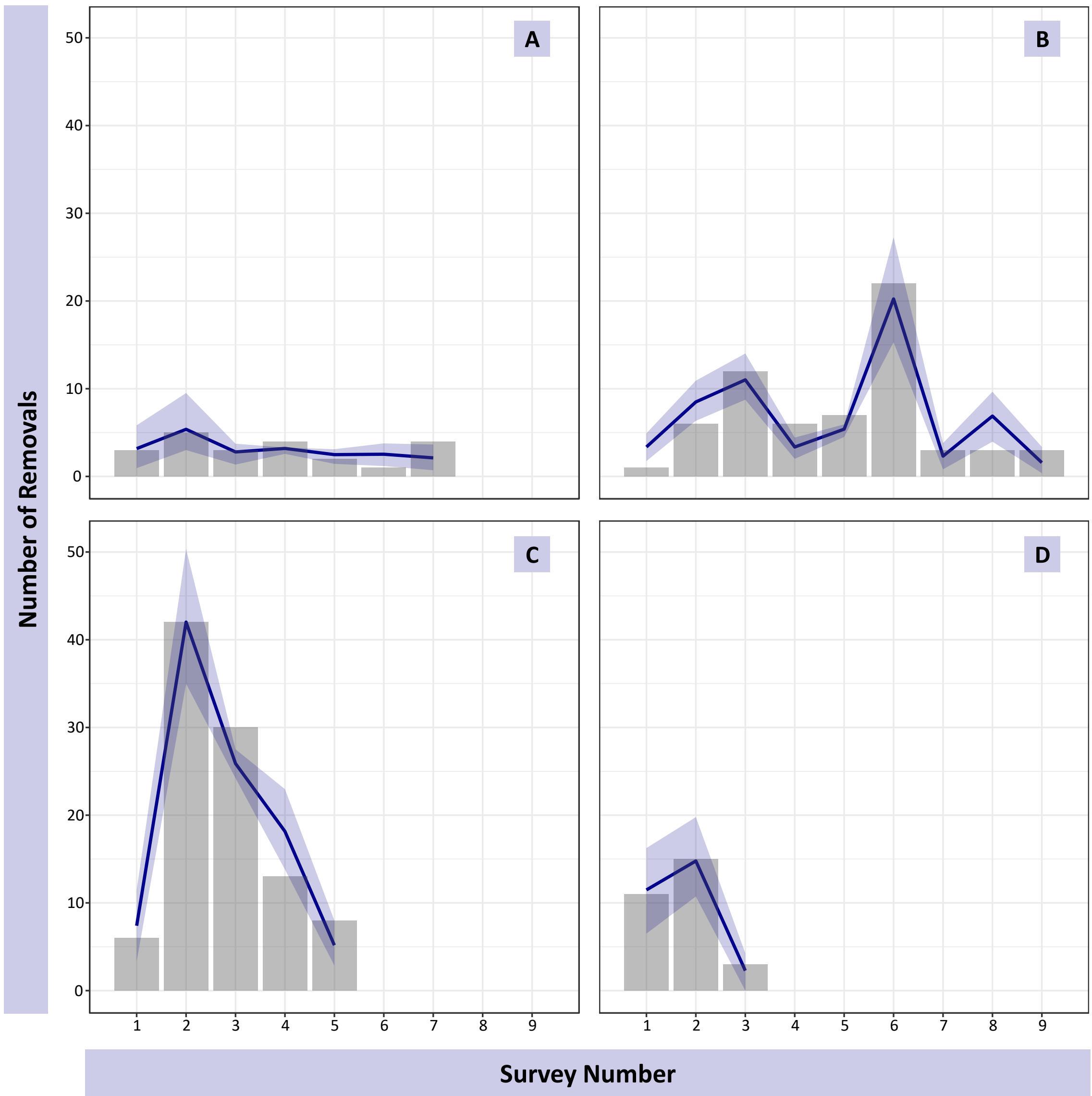

Figure 4.4: Number of Bojer's skinks removed from each release enclosure prior to the translocation. Grey bars represent the number of skinks captured for each survey and blue lines show the geometric removal model estimates of animals removed, with 95% confidence intervals given by the blue shaded areas around the line. Survey numbers account for the number of removal occasions for each enclosure.

4.4.3 Survival

Estimates of monthly survival ϕ varied most by release enclosure and sex, with our largest enclosure, enclosure C, having the highest ϕ (maximum likelihood estimate MLE: female = 0.897, male = 0.918) and the remaining enclosures having lower values (MLE: f = 0.747-0.780, m = 0.792-0.820; Figure 4.5). Male ϕ was higher than female in all enclosures, by approximately 0.06 on average (Figure 4.5). The highest ranked model for *p* was $p{\sim}effort + substrate\ temperature + site + sex$. The estimates of *p* were highest for enclosures A and D, lowest for enclosure C, and consistently higher for females (Figure 4.6). However, there were 38 models within 2 AICc of the top model, across which all environmental variables used were present in addition to enclosure and sex. To account for this model uncertainty, we used model averaging (Buckland *et al.* 1997) and compared the estimates to those from our highest ranked model (4.6.4 Supplementary Information: Table S4.4). In addition, some of the top ranked models ϕ were time dependent, but gave boundary estimates due to small numbers of observed individuals. Estimates of parameters were robust to model choice – we compared estimates from the averaged models, averaged models without those that gave boundary estimates and our top ranked model and found no appreciable difference between estimates (Figures 4.7-4.8). Confidence intervals were substantially larger than those for the top model when including the averaged models with boundary estimates but very similar when those were excluded. This implies that although ϕ consistently varies with enclosure and sex, *p* is likely due to a combination of environmental variables, in addition to enclosure and sex, which is too complex to explain with our current covariates.

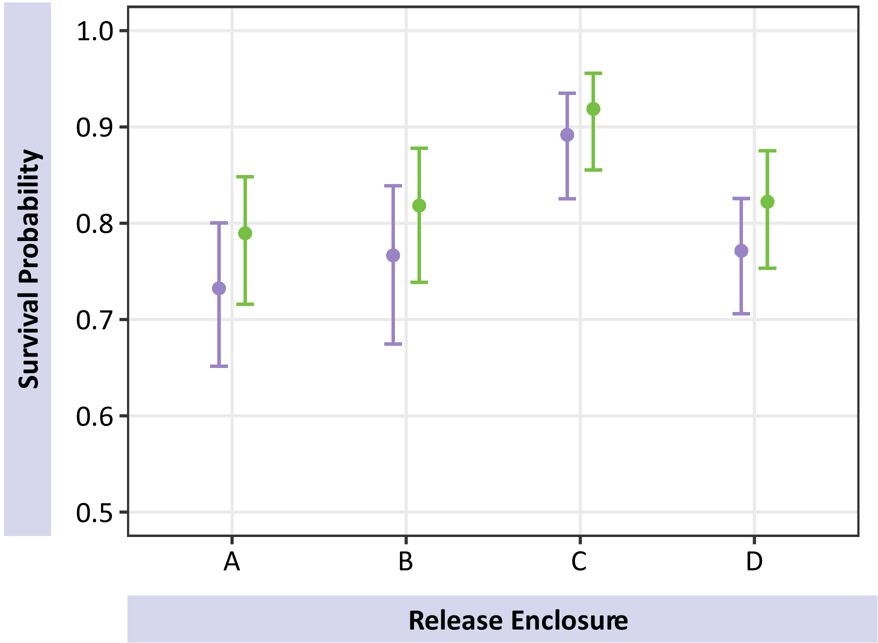

Figure 4.5: Variation in survival probability of the release animals by release enclosure and sex (male: green; female: purple) estimated by our highest ranked CJS model: $\phi \sim enclosure + sex, p \sim survey\ effort + substrate\ temperature + enclosure + sex$. Error bars show 95% confidence intervals, survival is estimated for the time between release, November 2022, and May 2023.

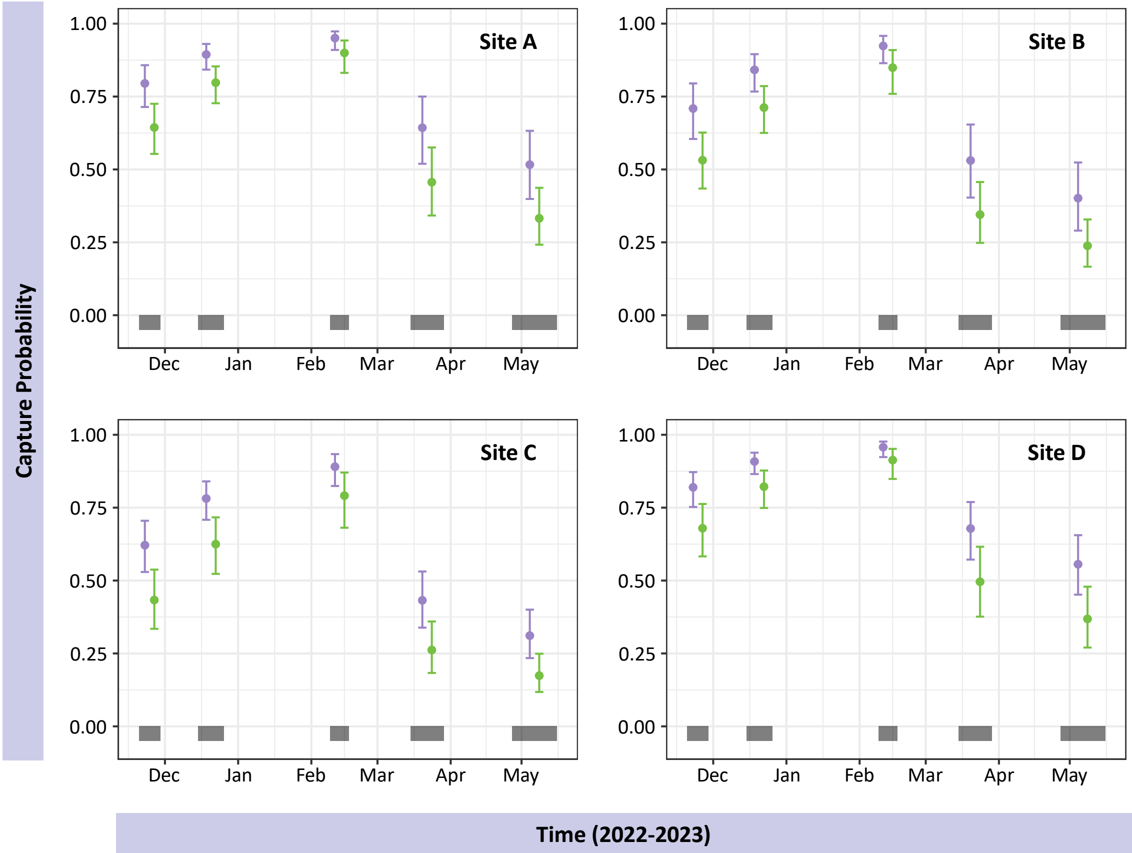

Figure 4.6: Variation in capture probability for release animals by enclosure and sex (male: green; female: purple) from the top ranked CJS model $\phi \sim enclosure + sex, p \sim survey\ effort + substrate\ temperature + enclosure + sex$. Capture probability is given for each post-release survey, with survey periods being indicated by the grey markers at the bottom of each panel. Error bars represent 95% confidence intervals.

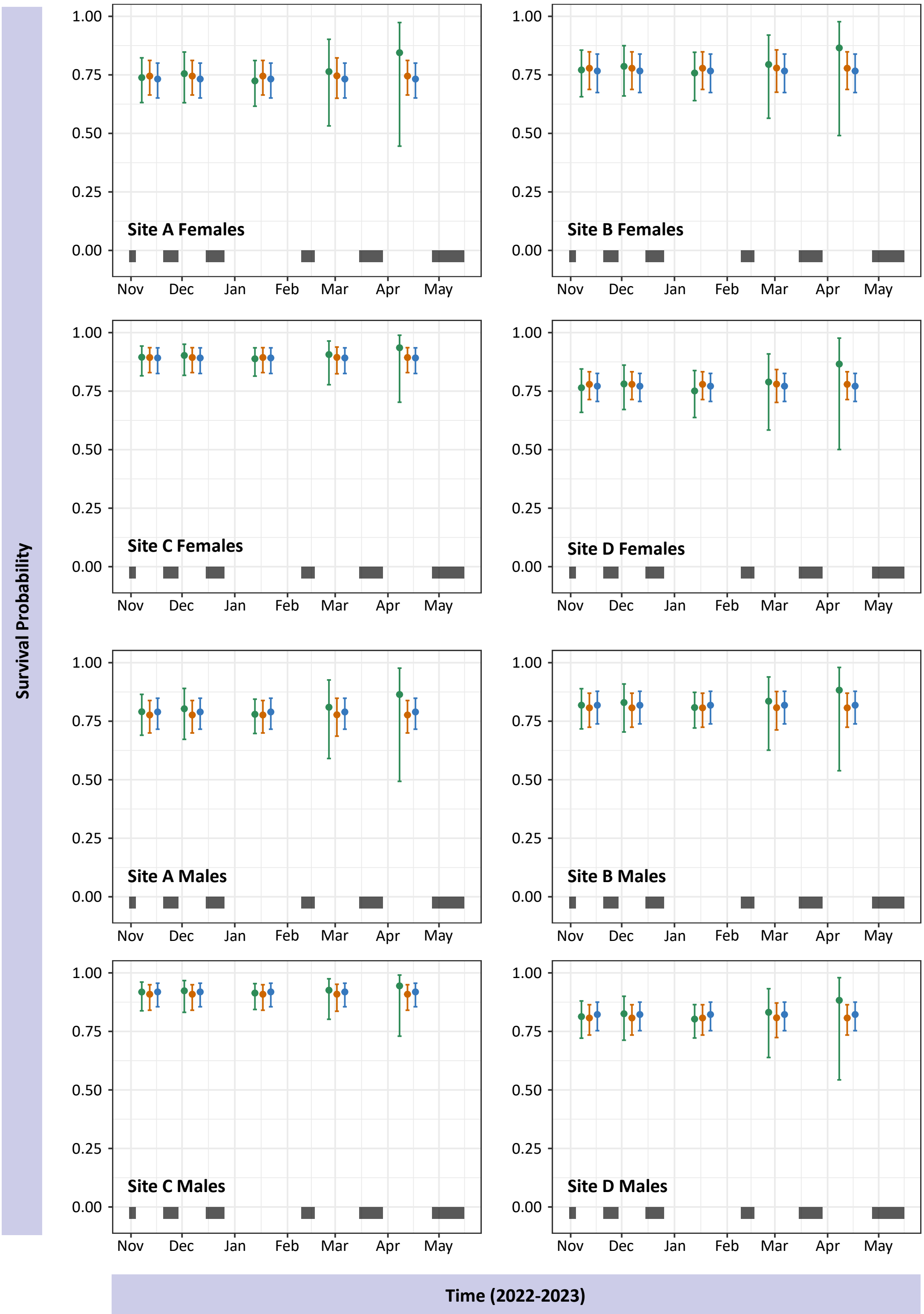

Figure 4.7: Comparison of estimates of survival probability for each enclosure from the top ranking CJS model (blue), the top ranked CJS models with dAICc under 2, averaged including those with boundary estimates (green) and excluding those with boundary estimates (orange). Error bars represent 95% confidence intervals and grey bars at the bottom show survey periods.

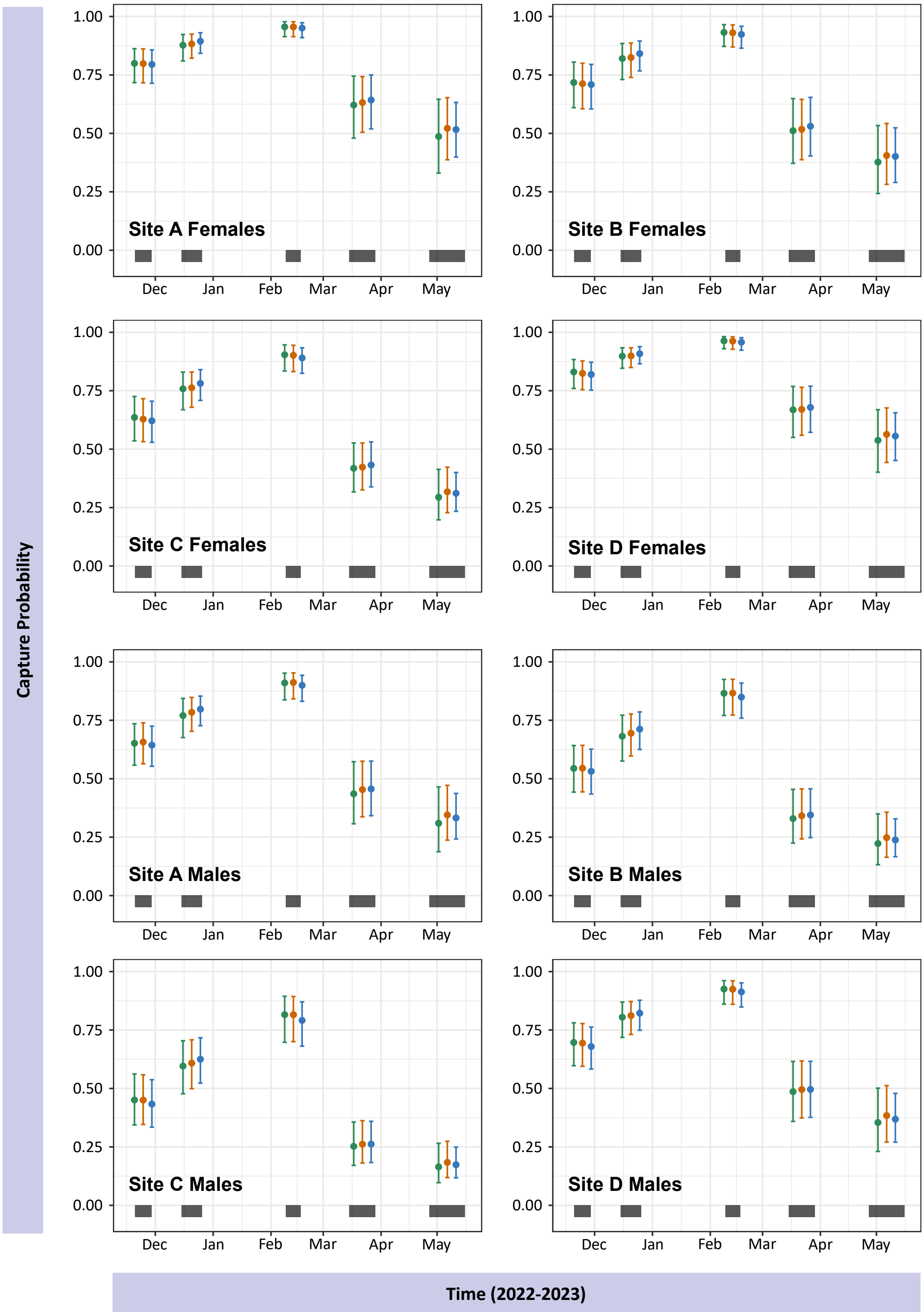

Figure 4.8: Comparison of estimates of capture probability for each enclosure from the top ranking CJS model (blue), the top ranked CJS models (with dAICc under 2) averaged including those with boundary estimates (green) and excluding those with boundary estimates (orange). Error bars represent 95% confidence intervals and grey bars at the bottom show survey periods.

4.4.4 Dispersal

The scaled MCP values show an initial increase in population distribution between release and the first survey in all enclosures, especially in the largest enclosure, C (Figure 4.9). The change in scaled area use then varies between enclosures, decreasing consistently in enclosure C. In enclosure A, the smallest enclosure, it remains constant then decrease at the final survey. Enclosures B and D show greater variability in area use relative to the number of captures. However, MCPs are only sensitive to the outermost boundaries of a species range therefore may not be a true representation of changes in dispersal.

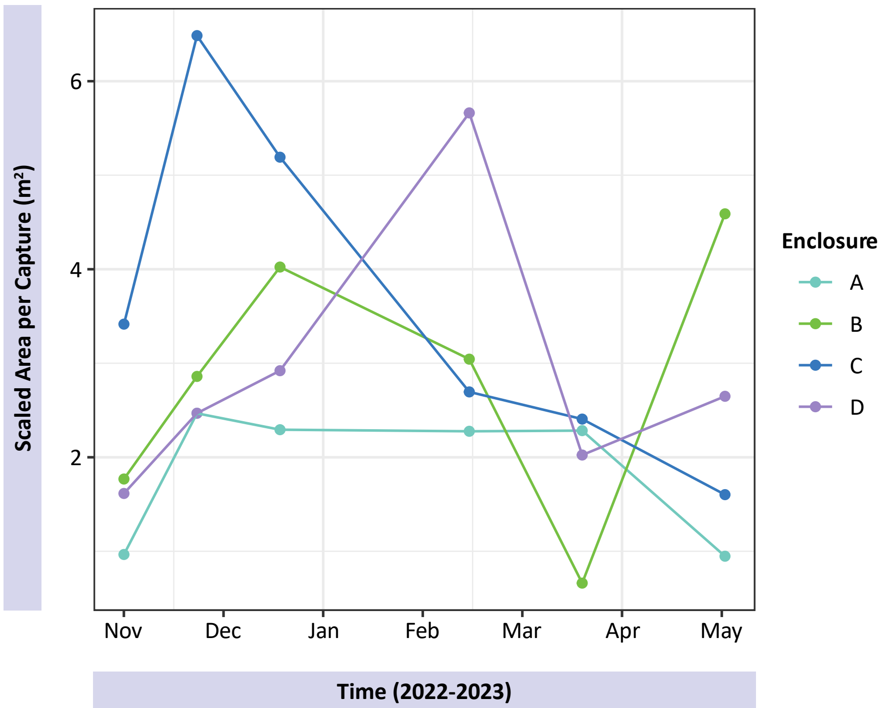

Figure 4.9: Temporal variation in minimum convex polygon area for each lesser night gecko release enclosure over the seven months between release and the fifth capture-recapture survey, scaled by number of captures per survey.

Cumulative distance travelled by each individual, in the cases where individuals were caught more than 3 times, varied in pattern. In some cases, greater distances were moved initially, then became more sedentary as predicted, variation in distance travelled showed no temporal trend. Although we do not have enough captures to test this statistically, there appears to be no difference between male and female dispersal (4.6.5 Supplementary Information Figures S4.5-S4.8). Total distance travelled by each individual was slightly higher in larger enclosures but not significantly (ANOVA: $F = 1.11$, $p = 0.296$).

4.5 Discussion

Seven months after release of lesser night geckos to Round Island, our results indicate that the use of soft-release enclosures for the reintroduction of lesser night geckos has been successful in reducing predation and dispersal. Our estimates of survival give a monthly probability of 0.75-0.92, indicating the majority of the translocated animals have survived, suggesting that both the removal, suppression and exclusion of predators was at least partly successful. Although low recapture rates do not allow further statistical analysis, enclosures also likely reduced post-release dispersal, as individuals in all enclosures were observed moving across the entirety of the enclosure, which could have resulted in a movement further from the release enclosures if the fences had not been in place.

In our initial planning review, we found that translocations of reptile species that are prey items for a native species remain uncommon and rarely published, but in New Zealand there have been several reptile translocations where invasive predators are present (Lettink *et al.* 2010; Bogisch *et al.* 2016; Romijn & Hartley 2016). We therefore found value in formally collecting expert opinion, in addition to published literature on a more diverse range of taxa, to inform our planning. Expert elicitation has been used in the conservation of many taxa to inform decisions where data is insufficient to support decision making or to quantify uncertainty (Runge *et al.* 2011; Martin *et al.* 2012; Bolam *et al.* 2019). The recommendations from the experts echoed literature from other reintroductions of prey species where soft-release enclosures increased success rates, especially with reptiles (Tetzlaff *et al.* 2019; Resende *et al.* 2021), and raised concerns that may not be represented in the published literature, due to the bias against publication of failure. Soft-release enclosures are also recommended from a welfare perspective and are thought to reduce stress induced by predation (Teixeira *et al.* 2007). Given uncertainty in the literature and expert recommendations, ideally we would have used an experimental approach for the reintroduction, comparing a hard and soft approach to maximise learning. However, having identified predation and dispersal as major risks to success, releasing a cryptic threatened species into a predator rich environment with few natural barriers raised ethical and logistic issues. This risk is likely common elsewhere in endangered species management, particularly when working with risk-averse stakeholders (Tulloch *et al.* 2015; Canessa *et al.* 2020).

We successfully suppressed the population of Bojer's skinks within the release enclosures through our targeted removal surveys. We likely suppressed the populations of Telfair's skink, Gunther's gecko and keel-scaled boa to at least the same level as the Bojer's skink. Telfair's skinks have a higher detection probability (Cole *et al.* 2018c) than both Bojer's skinks and lesser night geckos, as they are larger in size (Pernetta *et al.* 2005) and generally curious of human activity. Gunther's geckos, although more elusive than Telfair's skinks, are still much larger than Bojer's skinks and are predominantly arboreal; the habitat within the enclosures is likely not optimal for these species, as there was only one suitable palm in one of the four enclosures (in which we found the only adult Gunther's gecko detected). The other Gunther's gecko removal was a juvenile found on the fence of enclosure C, in which there was no suitable habitat, suggesting this individual might have been moving between habitat patches, not remaining in the enclosure regularly. There is greater uncertainty about whether the keel-scaled boas were removed, as they are present in all habitats on Round Island; however, night surveys were undertaken at least every other night for each enclosure in the two weeks prior to translocation and none were detected inside of the enclosures or within a 5-m radius of any enclosure. Once fences are made leaky (adding holes to allow dispersal of lesser night geckos away from the enclosure) it is possible Bojer's skinks and juvenile predatory reptiles will re-enter the enclosures. However, by providing a predator-free environment for the initial post-release period, lesser night geckos may have had enough time to find suitable refugia and to begin breeding, making the populations larger and less vulnerable to predation. This has been demonstrated in the conservation of *Oligosoma maccanni* in New Zealand, where the exclusion of non-native predators using an exclusion fence increased survival relative to a control site (Lettink *et al.* 2010).

The estimates of survival probability (0.75-0.92) from our CJS models indicate that the enclosures have allowed the majority of lesser night gecko individuals to persist following the translocation. It is likely that survival of the translocated animals was improved by the enclosures due to a decreased predation rate following the initial removal of predators and the exclusion of them throughout the early stages of post-release establishment. This has been demonstrated in reptilian prey species, although most commonly with non-native predators (Lettink *et al.* 2010; Smith *et al.* 2013; Knox & Monks 2014; Tanentzap & Lloyd 2017; Stokeld *et al.* 2018). Survival was highest in the largest enclosure C and decreased with enclosure size, possibly because greater resource availability reduced competition between translocated animals for food or refugia, or due to variation in habitat suitability across four enclosures. Enclosure C also had the most variable habitat due to the large size therefore the heterogeneity could have allowed for higher true survival or more variability in individual capture probabilities between individuals, depending on where they chose to settle within enclosures, which violates the assumption of equal capture rates and may have artificially inflated the survival probability. Estimates of male survival were higher when compared to females, likely due to their smaller size allowing them to use refugia which is inaccessible to most predators. Conversely, females had a higher detection probability, likely due to their larger size (Bullock *et al.* 1985). Additionally, many females translocated were gravid or became gravid shortly after release which also increased the likelihood of detection due to their increased size and slower movements**.** Detection probability of the released animals was initially higher than in the initial post-release period of the Ile Marianne population (see Chapter 2). This could be due to post-release stress and competition for resources causing individuals to move more around enclosures (Dickens *et al.* 2010), leaving them more exposed to capture. This is supported by our capture-recapture data, where a proportion of geckos are caught around the edges of the enclosures, on soil or loose rocks, in the first three months post-release, whereas no individuals were caught or sighted on soil and very few on loose rocks during later surveys. Environmental factors, as demonstrated by our CJS models, play a role in detection probability but are too interlinked to be able to determine the exact combination of environmental factors explaining the detection probability estimates. The decrease in detection probability with time is potentially due to a change in surveyor, with less experience, for the 4th and 5th surveys and a seasonal increase in vegetation cover for the final two surveys, although it may also reflect behavioural changes following the establishment of home ranges.

The lower numbers of captures in the March and May surveys, along with few individuals being recaptured in each survey period, reduced our ability to carry out quantitative analysis of post-release dispersal within the enclosures. However, we were at least able to demonstrate an increase in range for the total population between release and following two surveys. The occurrence of individual movements across the entire width of the enclosures suggests that the fences are limiting dispersal, as if individuals were able to disperse the same distance outside of enclosures, it is less likely that, once established, home ranges would overlap, leading to lower rates of reproduction and a reduced chance of population persistence, as was demonstrated in jewelled geckos (Knox & Monks, 2014). Additionally, with the enclosures being left in place for seven months, the chances of home ranges establishing within or close to the release enclosures is more likely (de Milliano *et al.* 2016, Knox *et al.* 2017; Jensen *et al.* 2021), which will aid persistence as lesser night geckos have home ranges of approximately 3 m^2 so the initial population must be within a small area to enable reproduction. In future, we plan to apply kernel-based methods for estimating dispersal as this has been shown to be more robust at low sample sizes than minimum convex polygons (Börger *et al.* 2006). Additionally, we plan to analyse the data collected on displacement from release location using net squared displacement modelling (Börger & Fryxell *et al.* 2012), however both analyses were outside the scope of this study given the time constraints and data available at the time of analysis.

Short-term estimates of survival during the establishment phase often differ from long-term survival trends, especially in longer lived and slow to mature reptile species (Bertolero *et al.* 2018). Therefore, continued long-term monitoring is key to assessing the success of reintroductions (Germano &

Bishop 2009; Bubac *et al.* 2019) and to obtaining estimates of survival for use in long-term projection models that can guide ongoing and future management. The results for our removal, survival and dispersal analyses all illustrate the difficulties in monitoring cryptic threatened prey species. Both lesser night gecko and Bojer's skink typically have low capture probabilities leading to small sample sizes of data, especially over short survey periods. This scenario leads to greater uncertainty in estimates of demographic parameters and therefore less certainty of success which is historically less likely with smaller release populations (Germano & Bishop 2009). Furthermore, there are inherent difficulties associated with small populations, which can be unavoidable when translocating threatened species, such as Allee effects and genetic drift (Germano & Bishop 2009). Lastly, post-release stress can lead to changes in behaviour and difficulty finding mates, decreasing reproductive output (Dickens *et al.* 2010). With the combination of these factors, as well as risks of predation and dispersal, the lesser night gecko translocation presented many challenges. However, the data collected so far, and the presence of gravid females and juveniles, suggests we can be confident of at least partial success of the initial establishment stage. The next stages of management, making fences leaky then removing them completely (detailed plan in 4.6.6 Supplementary Information), are likely to increase predation and dispersal, which could be mitigated by use of additional artificial refugia to connect the release enclosures with patches of high quality habitat. These factors may pose risks to a small population, but are also natural dynamics of a healthy population, which is the ultimate goal of the reintroduction. Through continued intensive monitoring we will seek to capture such population dynamics, and to make informed decisions about needs for future management, including additional releases where necessary.

4.6 Supplementary Information

4.6.1 Survey design

Lesser Night Gecko Translocation

What is the problem?

The lesser night gecko (hereafter LNG) is one of many threatened endemic reptiles found in Mauritius, classified as "Vulnerable" by the IUCN RedList. The LNG was probably widespread across Mauritius, but invasive mammals and reptiles have caused severe declines. The LNG is now found only on four outlying islands (see map below), three of which are smaller than 2 ha. Strict biosecurity measures are in place but the risk of invasion is still high. Extending the range of the species by translocation to other invasive-free islands is key to its persistence.

Lesser night gecko (*Nactus coindemirensis*)

What do we need to decide?

We plan to translocate individuals from the largest LNG population (~18000 individuals on Gunner's Quoin)to the neighbouring Round Island. Round Island has been the subject of a long-term restoration project. Its habitat and biotic community are similar to Gunner's Quoin. However, Round Island also contains three native predators – the Round Island boa, Telfair's skink and Guenther's gecko. Translocating LNG into an abundant predator system affects our choice of release method. We are uncertain about the behaviour of both LNG and predators, and about the overall ecosystem effects.

Round Island Boa
Casarea dussumieri
max. 150cm
Round Island to Gunner's Quoin Translocation 2012

Guenther's Gecko
Phelsuma guentheri
max. 28cm

Telfair's Skink
Leiolopisma telfairii
max. 30-40cm
Round Island to Gunner's Quoin Translocation 2007

Lesser Night Gecko
Nactus coindemirensis
max. 6cm
Gunner's Quoin to Round Island Translocation planned 2020

Who is involved?

Translocation specialists within the Durrell Wildlife Conservation Trust, Mauritian Wildlife Foundation and Mauritian Government will develop a conservation translocation plan, in association with the Zoological Society of London and the University of Kent, UK. A range of approaches and projected outcomes will be presented to the Mauritian Government in April 2020, and a plan agreed, with an aim to carry out the first translocation by September 2020.

Objective

Minimise the loss of translocated individuals to predation at the release site.

Photo credits: Dr Nik Cole

Figure S4.1: Study system information provided to interviewees before surveys commenced at the 9th World Congress of Herpetology.

Name:

Organisation:

Contact details:

Translocation Experience: Conservation Mitigation Both

1) Assuming there are no other concerns, what release options would you propose for the project?

2) Could you rank these options (if more than one given)?

3) What are the associated risks with these methods, and are there any more general constraints we should consider during the planning process?

4) Would you be happy to be contacted again in the future regarding this project?
   Yes □ No □

5) Do we have your permission to include your name on a list of experts consulted in a brainstorming exercise aimed at generating a list of possible release options to, not specifically linked to your answers, when presenting our plan to the National Parks and Conservation Service in Mauritius?
   Yes □ No □

***Figure S4.2*:** Questionnaire used for expert interviews at the 9th World Congress of Herpetology.

4.6.2 Survey protocol & release enclosure habitat maps

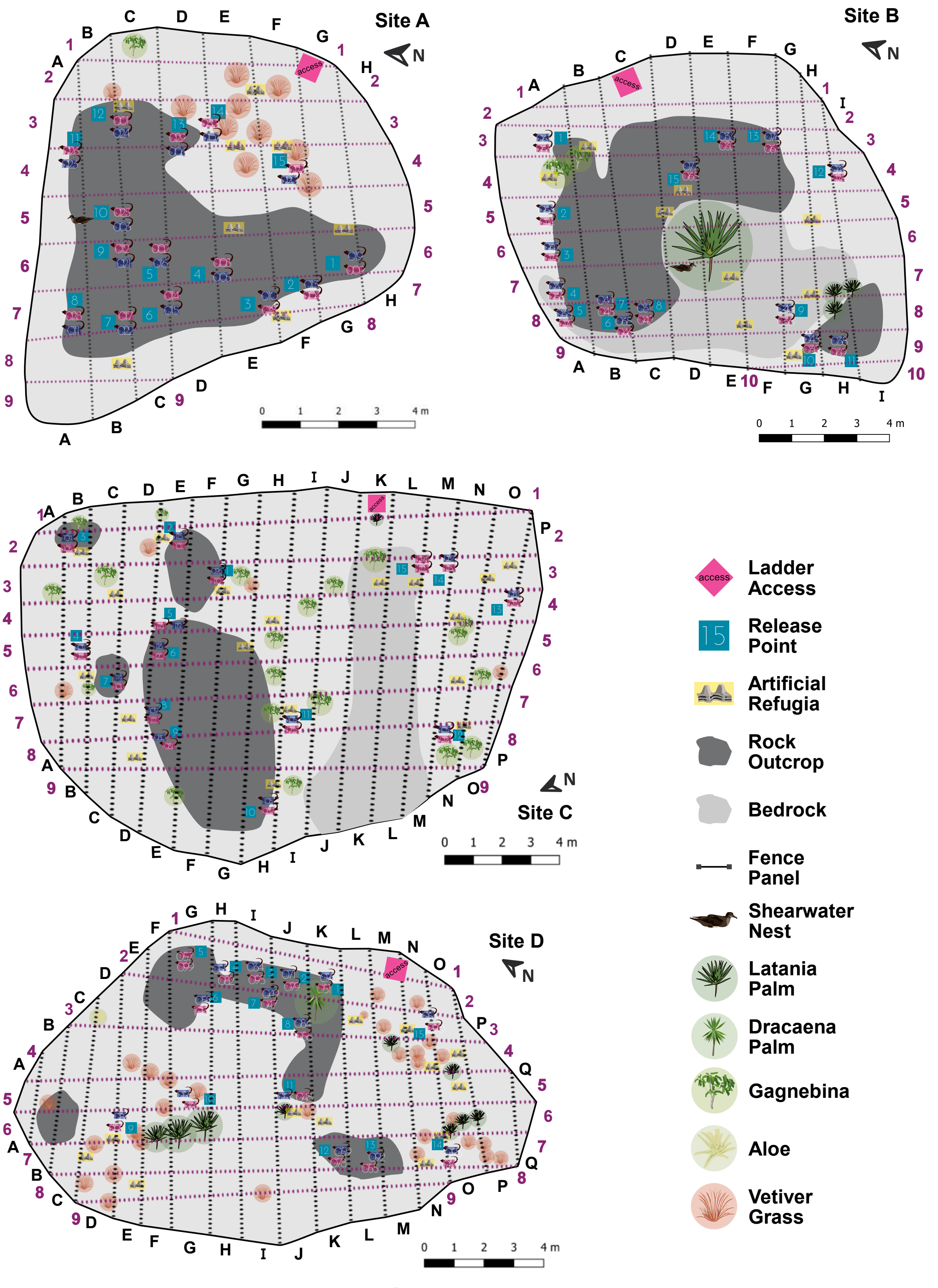

Figure S4.3: Schematic map of each release enclosure, with grid reference system and key points marked. Each gecko represents the individual released with ID number, (male: blue, female: pink).

Lesser Night Gecko Survey Protocol:

Capture-mark-recapture surveys commenced at the end of November 2022, approximately 3 weeks after release into the enclosures. Two sets of surveys were carried out (one in November and one in December) by Katie Bickerton with assistance from the Round Island wardens and Durrell MIRI interns. Surveys are to be continued at 4–6 week intervals, with a full set of surveys consisting of 3-4 surveys of each enclosure over a 2 week period (weather dependent).

Survey Protocol

Surveys should be carried out once completely dark and preferable avoiding weeks where there is a full moon. While enclosures remain closed, two enclosures should be surveyed each night (ideally enclosures A and C followed by enclosures B and D) and alternated until 3-4 surveys of each enclosure have been carried out. Each enclosure should be searched once following the pathways shown in Figure S4.4 – these are specific to habitat within each enclosure and can be modified slightly where more than one person is monitoring. Each survey should take between 40 minutes – 1 hour dependent on how many geckos are found.

Before leaving the field station, the most recent version of the data sheet should be downloaded and saved on the field phone with the days date. Ensure you have all items of field kit for the surveys:

- Field Phone (charged with most recent datasheet downloaded)
- Headtorch & spare batteries
- Callipers
- Scales
- Plastic reptile bags
- Camera
- Cloth reptile bag and notebook (for photo background)
- Utility belt (useful for carrying instruments when surveying alone)
- Radio

- First Aid Kit
- GPS (once holes are added to fences)

Once near the enclosure, record the date, species, island, enclosure, survey type, number of experienced staff, staff initials and initials of staff responsible. Also record the air temperature, cloud cover, moon phase, whether the moon is visible, wind and rain. Weigh an empty plastic reptile bag and record at the end of the data sheet. Details of all classifications are in the "Lists" tab of the data sheet. Ensure you have plastic reptile bags, callipers, scales, camera, white reptile bag for photos, thermometer and a radio.

When ready, set up the ladders, all habitat should be checked thoroughly before placing the ladder inside the enclosure at the access point in case of geckos close to that point, and the ladder should be tied to the fence using two ropes per ladder. Record the start time in the data sheet. Proceed along the survey path for the enclosure (Figure S4.4) checking around you before continuing and holding on to the fence where needed for stability, parts of the enclosures can be very slippery. Try not to shine your headtorch on to parts of the enclosure which have not been searched. When a gecko is seen, capture by hand, ensuring the animal is caught be the body not the tail. Once captured, immediately place the animal in the plastic reptile bag for processing. Record the substrate temperature, substrate type, whether the animal was moving when first seen and grid reference (using the maps saved on the data phone).

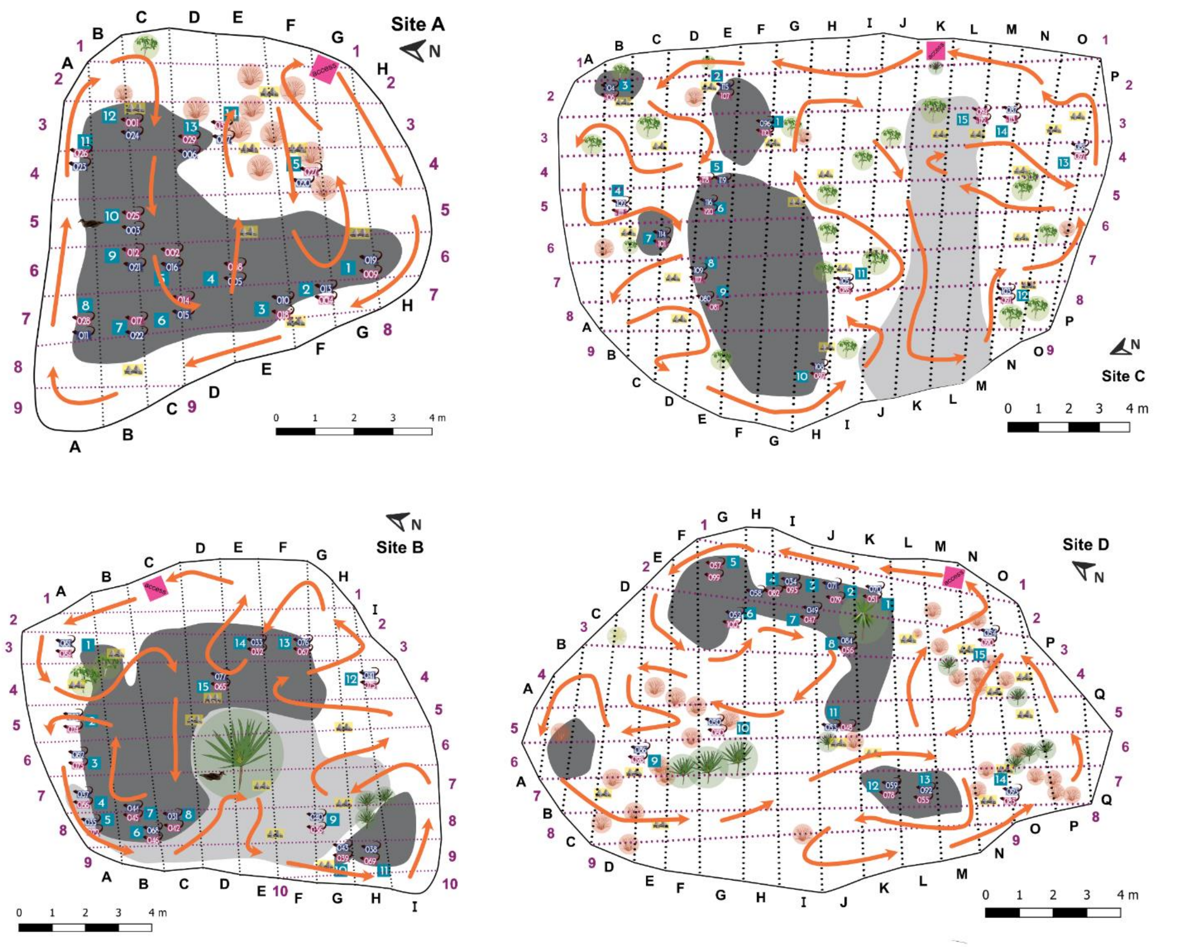

Figure S4.4: Survey routes for release enclosures A-D.

To process the gecko:

1. Sex and if female, check to see if it's gravid.
2. Check for scars, tail breaks and missing toes or claws (there should be 5 toes on each foot).
3. Weigh the gecko inside the bag and weigh the bag after if not done previously (the data sheet will calculate the weight for you).
4. Remove the gecko from the bag and measure the svl (snout-to-vent length) and tail length using the callipers.
5. Photograph the gecko by placing on a white background and pinning the back leg, all photos should be taken with flash, landscape with the head on the left and the body and tail straight. Each individual should be photographed twice.
6. Record number of photos taken and photo references.
7. Check all data has been recorded then release the gecko back to where it was caught.

4.6.3 Survey data

Table S4.1: Number of Bojer's skinks removed during targeted predator removal surveys for each enclosure, with details of number of staff and length of survey.

Enclosure	Date (am or pm)	Survey Length (minutes)	Number of Staff	Number Removed
A	24 Oct (pm)	20	1	3
	28 Oct (am)	70	2	8
	29 Oct (am)	60	2	3
	30 Oct (am)	35	3	4
	30 Oct (pm)	30	3	2
	31 Oct (am)	60	2	1
	31 Oct (pm)	60	2	4
B	24 Oct (pm)	30	1	1
	25 Oct (am)	60	2	6
	28 Oct (pm)	45	3	12
	29 Oct (am)	30	2	6
	29 Oct (pm)	55	2	7
	30 Oct (am)	90	3	22
	30 Oct (pm)	30	3	3
	01 Nov (am)	70	3	3
	02 Nov (am)	25	1	3
C	25 Oct (am)	60	2	6
	01 Nov (pm)	105	3	42
	03 Nov (pm)	102	3	30
	04 Nov (am)	155	2	13
	04 Nov (pm)	105	2	8
D	26 Oct (pm)	75	2	11
	02 Nov (pm)	140	3	15
	03 Nov (am)	60	3	3

Table S4.2: Number of predators removed from each release enclosure before the lesser night gecko translocation.

Species	Number Removed from Enclosure				
	A	B	C	D	Total
Telfair's Skink	9	9	7	1	26
Bojer's Skink	25	63	99	33	220
Gunther's Gecko	0	1	1	0	2
Keel-Scaled Boa	0	0	0	0	0
Total	34	73	107	34	248

Table S4.3: Mark-recapture surveys per enclosure for each survey trip and lead surveyor (either Katie Bickerton, Rouben Mootoocurpen or Alex Ferguson).

Trip	Trip Dates		Surveys per Enclosure				Lead Surveyor
	Start	End	A	B	C	D	
Nov 22	20-11-2022	29-11-2022	4	4	4	3	Katie
Dec 22	15-12-2022	26-12-2022	4	4	4	5	Katie
Feb 23	09-02-2023	17-02-2023	4	3	4	3	Rouben
Mar 23	13-03-2023	29-03-2023	4	3	3	3	Alex
May 23	26-04-2023	16-05-2023	4	3	3	4	Alex

4.6.4 Survival model selection & covariate information

Table S4.4: Top ranked CJS models by within 2 ΔAICc of the top ranked model from the survival analysis with survival and capture probability covariates listed, where n is the number of model parameters. Models marked with * were excluded from the second model averaging method due to boundary estimates. Bottom 3 models (with constant or time dependent survival and capture are included for completeness and were not used in the model averaging process).

Survival	Capture	n	AICc	ΔAICc	weight
enclosure + sex	effort + substrate temperature + enclosure + sex	12	1786.22	0.000	0.031
*enclosure + sex + time	effort + humidity + cloud cover + enclosure + sex	17	1786.30	0.074	0.030
*enclosure + sex + time	surveyor + moon visibility + cloud cover + enclosure + sex	17	1786.31	0.083	0.030
*enclosure + sex + time	effort + moon visibility + cloud cover + enclosure + sex	17	1786.33	0.111	0.029
*enclosure + sex + time	moon visibility + humidity + cloud cover + enclosure + sex	17	1786.39	0.164	0.028
*enclosure + sex + time	moon phase + wind presence + cloud cover + enclosure + sex	17	1786.55	0.326	0.026
*enclosure + sex + time	substrate temperature + moon visibility + cloud cover + enclosure + sex	17	1786.56	0.334	0.026
*enclosure + sex + time	wind speed + rainfall + cloud cover + enclosure + sex	17	1786.58	0.360	0.026
enclosure	effort + substrate temperature + enclosure + sex	11	1786.67	0.445	0.025
*enclosure + sex + time	substrate temperature + humidity + cloud cover + enclosure + sex	17	1786.67	0.448	0.025
*enclosure + sex + time	surveyor + humidity + cloud cover + enclosure + sex	17	1786.77	0.550	0.023
enclosure + sex	substrate temperature + humidity + enclosure + sex	12	1786.87	0.643	0.022
*enclosure + sex + time	effort + substrate temperature + enclosure + sex	16	1786.92	0.702	0.022
enclosure	substrate temperature + humidity + enclosure + sex	11	1787.04	0.816	0.021
enclosure + sex	effort + humidity + cloud cover + enclosure + sex	13	1787.05	0.831	0.020
enclosure + sex	wind speed + rainfall + cloud cover + enclosure + sex	13	1787.06	0.841	0.020
enclosure + sex	effort + moon visibility + cloud cover + enclosure + sex	13	1787.07	0.851	0.020
enclosure + sex	moon visibility + humidity + cloud cover + enclosure + sex	13	1787.11	0.886	0.020

enclosure + sex	surveyor + moon visibility + cloud cover + enclosure + sex	13	1787.12	0.894	0.020
enclosure + sex	substrate temperature + moon visibility + cloud cover + enclosure + sex	13	1787.24	1.016	0.019
enclosure + sex	substrate temperature + humidity + cloud cover + enclosure + sex	13	1787.35	1.131	0.018
enclosure	effort + humidity + cloud cover + enclosure + sex	12	1787.39	1.169	0.017
enclosure	wind speed + rainfall + cloud cover + enclosure + sex	12	1787.39	1.172	0.017
*enclosure + sex + time	effort + substrate temperature + cloud cover + enclosure + sex	17	1787.40	1.179	0.017
enclosure	surveyor + moon visibility + cloud cover + enclosure + sex	12	1787.42	1.200	0.017
enclosure	effort + moon visibility + cloud cover + enclosure + sex	12	1787.43	1.203	0.017
enclosure	moon visibility + humidity + cloud cover + enclosure + sex	12	1787.47	1.251	0.017
*enclosure + sex + time	substrate temperature + moon phase + cloud cover + enclosure + sex	17	1787.49	1.269	0.016
enclosure	substrate temperature + moon visibility + cloud cover + enclosure + sex	12	1787.63	1.409	0.015
enclosure	substrate temperature + humidity + cloud cover + enclosure + sex	12	1787.76	1.540	0.014
enclosure + sex	surveyor + humidity + cloud cover + enclosure + sex	13	1787.78	1.562	0.014
enclosure + sex	moon phase + wind presence + cloud cover + enclosure + sex	13	1787.98	1.756	0.013
enclosure	surveyor + humidity + cloud cover + enclosure + sex	12	1788.04	1.819	0.012
*enclosure + sex + time	wind speed + enclosure + sex	15	1788.13	1.906	0.012
enclosure + sex	effort + substrate temperature + cloud cover + enclosure + sex	13	1788.18	1.956	0.012
enclosure + sex + cyclone	effort + substrate temperature + enclosure + sex	13	1788.20	1.976	0.012
enclosure	moon phase + wind presence + cloud cover + enclosure + sex	12	1788.20	1.981	0.011
1	time	6	1819.76	33.532	0.000
time	1	6	1844.44	56.555	0.000
1	1	2	1908.80	58.214	0.000

Table S4.5: Description of the time varying covariates used in the survival models, with the units used and range of values. All values were scaled for use in survival models.

Covariate	**Description**	**Units**	**Range**
Effort	Mean number of surveys carried out per enclosure within each survey period.	No. of surveys	3.25 – 4
Surveyor	A scaled variable used to indicate who was surveying the enclosure (three options within the survey period, 4.6.3 Supplementary Information Table S4.3).	NA	0 – 1
Substrate temperature	Temperature of the substrate where the individual was first sighted, averaged across all captures.	°C	24.5 – 27.6
Moon phase	Moon phase expressed as a proportion of moon visible, where a new moon = 0 and a full moon = 1. Waxing and waning were not taken into account.	NA	0.20 – 0.51
Moon visibility	Visibility of moon at the time of survey, where not visible = 0, partially visible = 0.5 and fully visible = 1, averaged across all surveys.	NA	0 – 0.33
Wind presence	A qualitative measure of wind as a true measure of wind speed was not available for each enclosure, where no wind = 0, mild (that could be felt on the skin) = 0.5, windy (where clothes moved) = 1.	NA	0.23 – 0.75
Wind speed	Mean wind speed during each survey period, from the weather station at the field station.	m/s	3.07 – 4.09
Humidity	Mean relative humidity recorded during the survey period from the weather station at the field station.	%	78.9 – 87.4
Rainfall	Mean rainfall during the survey period measured by the weather station at the field station.	mm	0 – 0.018
Cloud cover	Mean cloud cover during each survey period, estimated by surveyor to the nearest 10%.	%	20.5 – 41.0
Cyclone	A binary covariate used to measure the impact of a cyclone between the February and March surveys on survival, where 1 was used to represent the cyclone.	–A	0 - 1

4.6.5 Individual dispersal figures

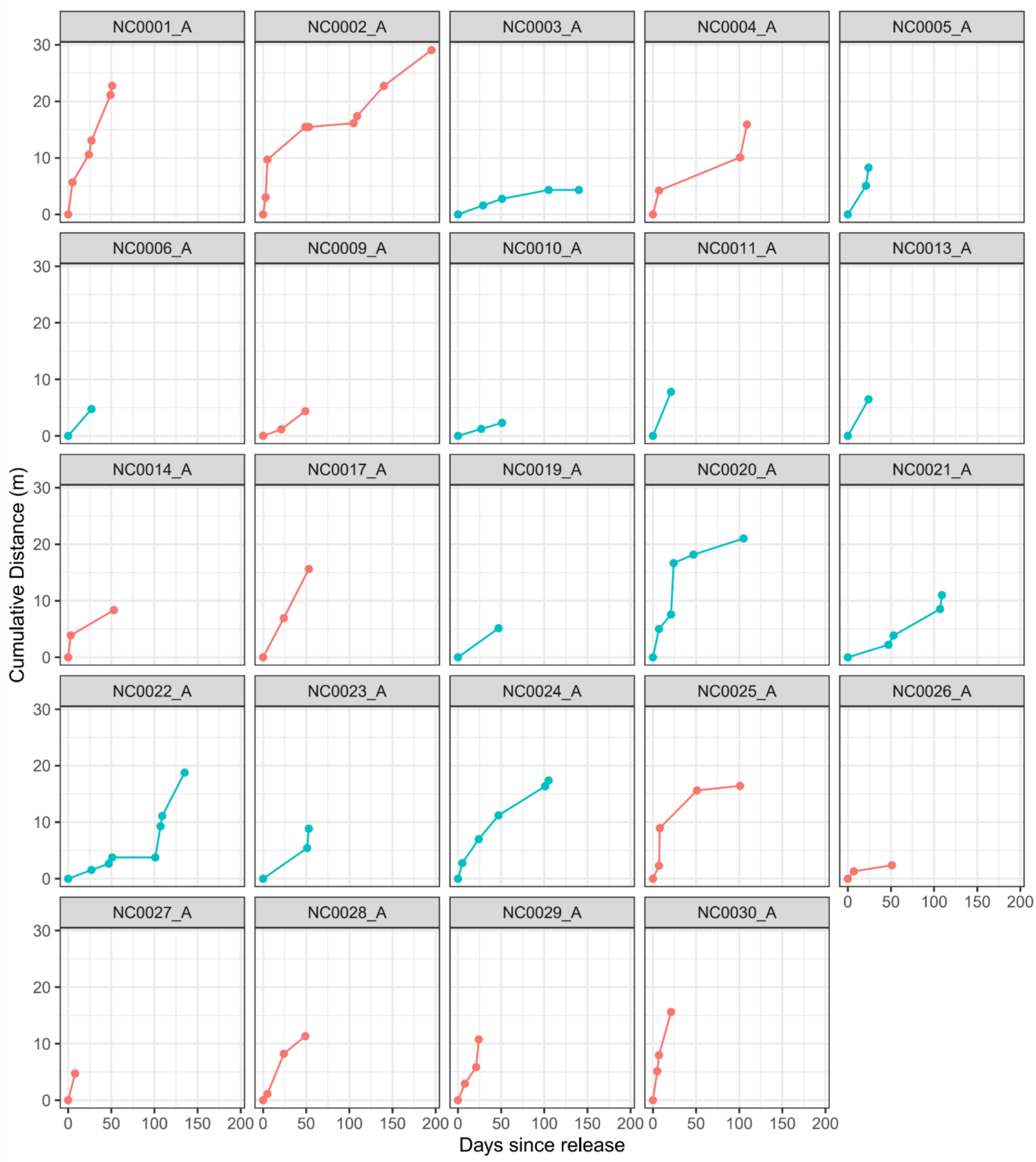

Figure S4.5: Movements of individual geckos in enclosure A shown as cumulative distance moved, in meters, from release location in the days since release. Points mark capture events, red: female, blue: male. Individuals that have not been captured since release are not shown.

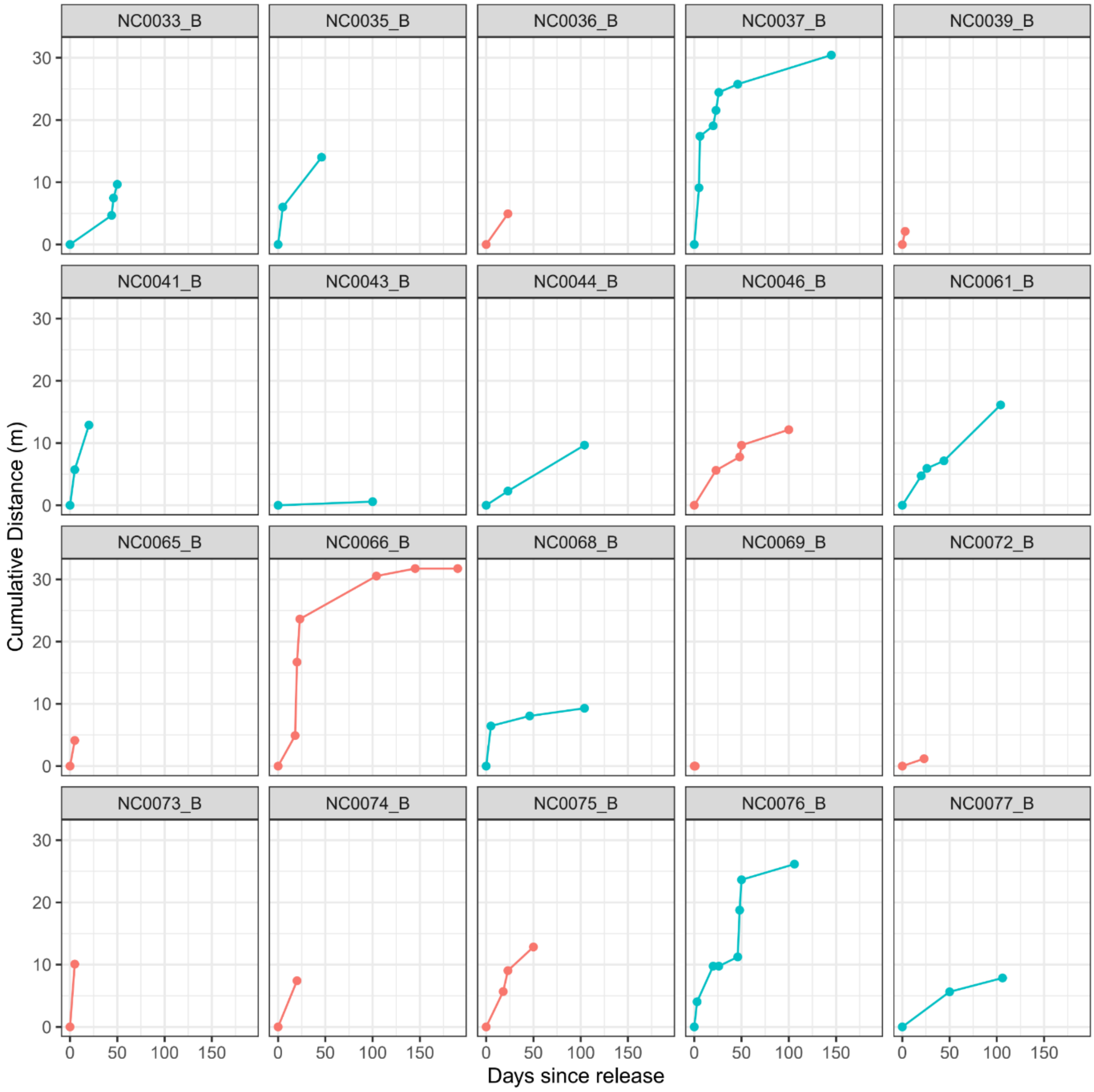

Figure S4.6: Movements of individual geckos in enclosure B shown as cumulative distance moved, in meters, from release location in the days since release. Points mark capture events, red: female, blue: male. Individuals that have not been captured since release are not shown.

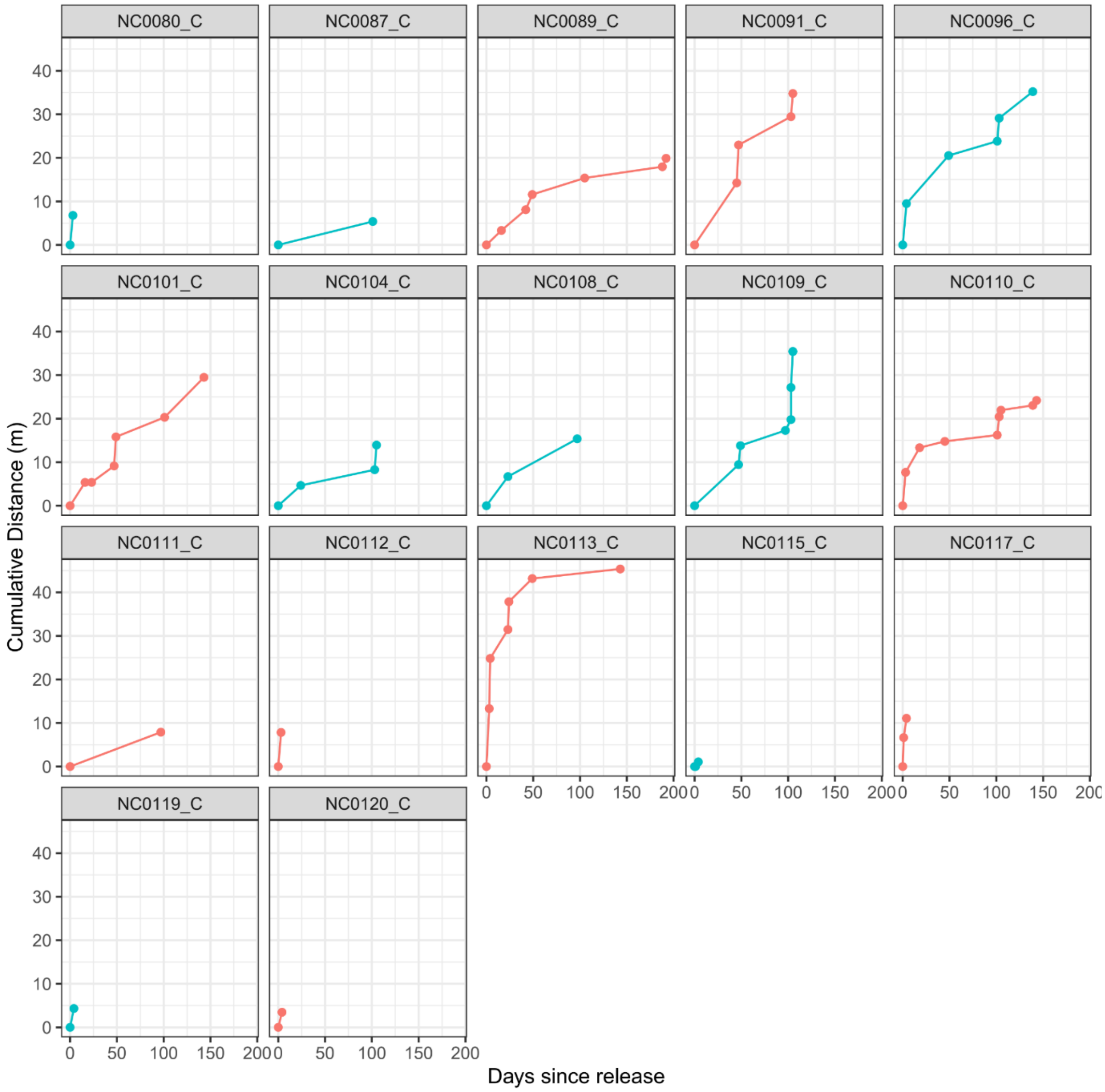

***Figure S4.7*: Movements of individual geckos in enclosure C shown as cumulative distance moved, in meters, from release location in the days since release. Points mark capture events, red: female, blue: male. Individuals that have not been captured since release are not shown.**

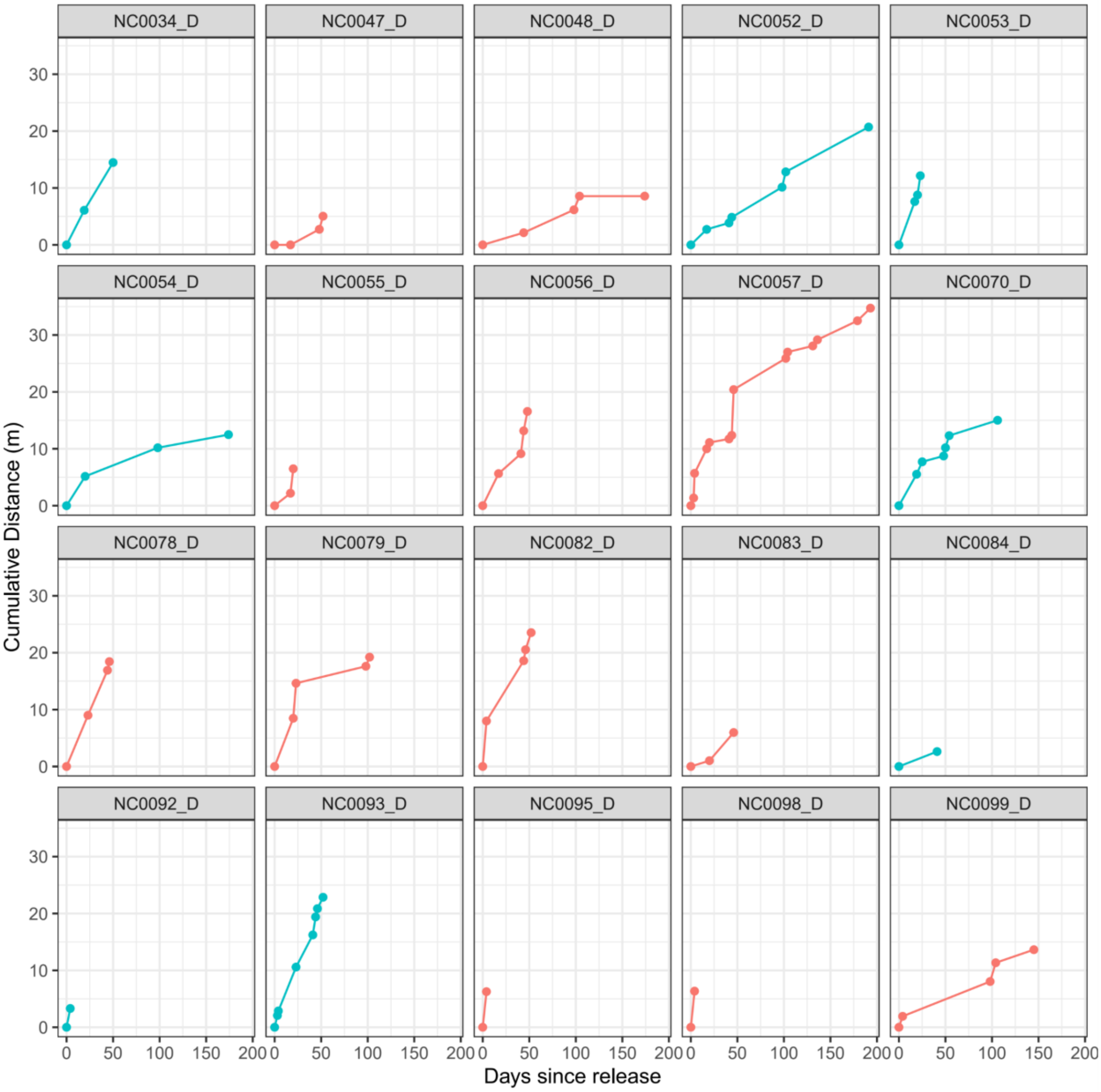

Figure S4.8: Movements of individual geckos in enclosure D shown as cumulative distance moved, in meters, from release location in the days since release. Points mark capture events, red: female, blue: male. Individuals that have not been captured since release are not shown.

4.6.6 Fence Removal Protocol

Fence Removal Plan

Fence removal will be determined by two factors: presence of juveniles and reduced dispersal. Once juveniles are present in the enclosure and as long as dispersal has reduced, the inner plastic can be removed – cut at ground level to prevent disturbing the rocks holding down the plastic which geckos may be using as refugia. The outer plastic should have holes about 1cmx0.5cm in size at ground level, approximately every 50cm (approx. 2-3 per fence panel). Following this, mark recapture monitoring should be extended to a 25m radius around each enclosure (Figure 7) in addition to within the enclosures. When full removal of plastic will occur is still yet to be decided but will be dependent on the success of the initial removal phase.

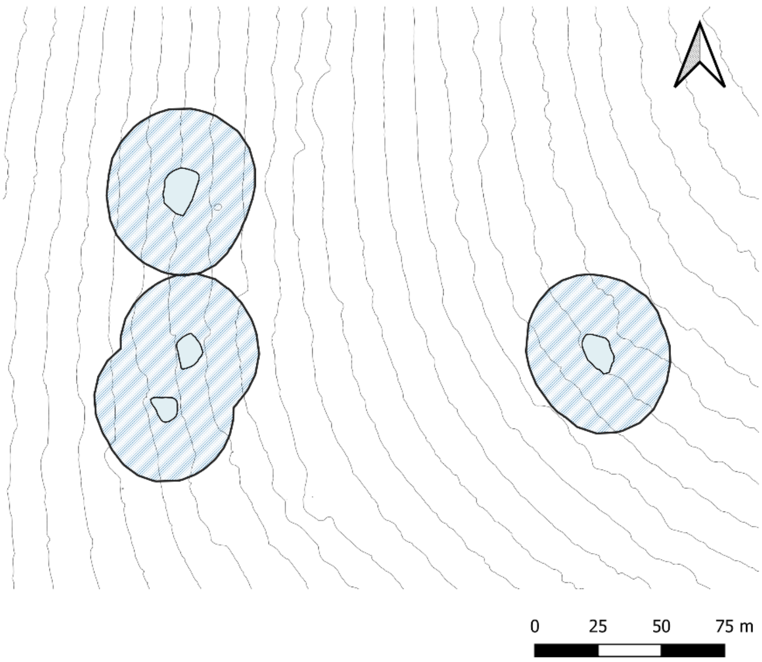

Figure S4.9: Extended survey areas.

5 Discussion

5.1 Summary of key findings

In this thesis, I explore the value of using bespoke capture-recapture models to understand the ecology and demographics of translocated reptile populations and to directly inform conservation decision-making. Reptiles are one of the most underrepresented vertebrate groups in the translocation literature (Germano & Bishop 2009; Bubac *et al.* 2019), therefore taxon-specific guidance for translocation planning is very limited. Furthermore, many reptile species exhibit highly cryptic behaviour (Towns *et al.* 2016; Bilby & Moseby 2023), limiting the species-specific ecological knowledge that is available for effective conservation planning, and complicating (re)capture after translocation. These challenges in turn make it difficult to determine success, especially in the initial establishment phase (Armstrong & Seddon 2008; IUCN/SSC 2013).

One of the most common methods of post-release monitoring are capture-recapture surveys (Dieterman *et al.* 2010; Dolny *et al.* 2018; Moseby *et al.* 2018; Aguirre *et al.* 2019), which enable surveyors to estimate key demographic parameters, such as population size, survival probability and probability of new entrants, which are often used to assess translocation success. Jolly-Seber (JS) models are frequently used to estimate these parameters; however, estimates often have high uncertainty in the early establishment phase prior to new entrants joining the population, providing limited insight for conservation management. In **Chapter 2**, I developed a novel modification to the POPAN formulation of the JS model to account for translocated individuals and the post-translocation delay in entry of new individuals to the population. Using a simulation study, I demonstrated that the modified model increased accuracy of parameter estimates in the initial post-release phase, especially in populations with low detection probability, and reduced the risk of overestimating abundance in these populations compared to using a standard JS model. I then fitted the modified model to a dataset from a reintroduction of a highly cryptic species of gecko, the lesser night gecko (*Nactus coindemirensis*), to Île Marianne, Mauritius. The results of this analysis confirmed insights from our simulation: the modified JS model is more accurate (having smaller confidence intervals) than the standard JS model, despite a very low number of recaptures in initial surveys.

In July 2020, the MV Wakashio ran aground on a reef 1.1 km from the coast of Mauritius, leaking into the surrounding reef system an estimated 1000 tonnes of oil, which then washed across several low-lying islands. Four of these islands, one of which was Île Marianne, host relict populations of two endemic Mauritian reptiles: lesser night geckos and Bojer's skinks (*Gongylomorphus bojerii*). This incident was an environmental disaster, however its timing during my PhD also presented an opportunity to address the important issue of what impact this catastrophe subsequently had on the endemic herpetofauna. Therefore, in **Chapter 3**, I examined the impact of the oil spill on the survival and body condition of the lesser night gecko and Bojer's skink across the four islands, using the modified JS model for the translocated populations and the standard POPAN formulation of the JS model for the non-translocated populations. Results suggested lower survival after the spill in the Île Marianne lesser night gecko population, although we cannot attribute this to the oil spill with certainty, due to large time periods between the surveys before and after the oil spill, as annual surveys (usually carried out in March or April) were cancelled in 2020 due to the Covid-19 pandemic. The other populations showed little variation in survival pre- and post-spill, likely because they were surveyed less often than the Île Marianne population, and therefore had smaller sample sizes. Decreases in body condition were detected in the Île Marianne lesser night gecko population and the Bojer's skink populations on Ile aux Fouquets and Ile de la Passe; however, due to the timing of the surveys, lower body condition could also be due to seasonal variation. The findings in this chapter emphasized the importance of baseline survey data and the difficulties faced in conservation decision making following an environmental disaster.

In **Chapter 4**, I undertook a translocation of lesser night geckos to Round Island, Mauritius. I carried out a semi-structured interview with experts in herpetofauna translocations to inform the initial planning of the translocation release and identify key risks. Following expert recommendations, I used soft-release enclosures to exclude predators and minimize dispersal of released animals. Prior to release, populations of the most numerous predators (including three species of lizards) were suppressed by an estimated 67-75%, assessed using geometric removal models accounting for time-varying detection probability using covariates of survey effort. Following release, capture-recapture surveys were used to monitor monthly survival probability for the first seven months post-release, which was estimated between 0.75-0.92 using a Cormack-Jolly-Seber model (the modified JS model described in Chapter 1 is not yet appropriate for the population as we do not have any new entrants into the adult population). The relatively high survival rate, presence of juveniles and considerable movement within enclosures, suggest that predation and dispersal have been reduced by the chosen release strategy.

In summary, the three data chapters give insights into the challenges faced when monitoring small populations of cryptic species and combine novel and existing statistical methods with ecological knowledge to improve conservation decision-making.

5.2 Comparison of Île Marianne and Round Island translocations – planning, release, monitoring & key findings

The planning, execution and outcomes of two translocations of lesser night geckos are discussed in this thesis: the Île Marianne translocation in 2011 and the Round Island translocation in 2022. The Île Marianne translocation of 75 individuals used a hard-release strategy (releasing animals directly into the release site). Hard-release was chosen because the island is small in area and has no predators, thus avoiding two key concerns for the Round Island translocation. Survival probabilities were broadly similar in the initial seven months of both translocations, with mean monthly survival across the seven-month post-release survey period estimated between 0.75-0.92 for Round Island (dependent on release site) and 0.94 for Île Marianne across the 10-year survey period. Detection probability was much higher on Round Island (mean 0.56 compared to 0.08 over the first seven months post-release), likely because released individuals were constrained to the release enclosures, reducing the survey area. Habitat also varies between the two islands, with more vegetation present on Ile Marianne, providing more refugia that could decrease detection.

The only environmental variable collected consistently across the 10-year survey period on Île Marianne was air temperature, whereas we consistently collected a range of environmental variables (including air temperature) on Round Island, in addition to having access to the island weather station. It is therefore unlikely the environmental factors influencing detection can be compared between the two islands, although temperature did affect detection in both populations. Despite differences in aims and metrics of success, the information from the Île Marianne translocation was useful in planning the Round Island translocation. For example, several individuals on Île Marianne were observed to move distances >80 m, whereas individuals born on the island were usually found within the same 5 m. This new information reinforced the initial assessment by experts of post-release dispersal as a risk to translocation success. The outcomes of the two translocations presented in this thesis demonstrate the variation between and within populations of the same species, demonstrating that previous translocations can be used to inform future translocations however difference in objectives, release strategies and release site habitat can limit transferability.

5.3 Contributions to the field: Statistics

Methods for statistical analysis increasingly focus on "big data" (Hampton *et al.* 2013; Farley *et al.* 2018; Niu *et al.* 2020; McCleery *et al.* 2023). However, conservation biologists working with threatened species are often faced with a very different situation of sparse and uncertain data from which to make management decisions (Todman *et al.* 2023). Conservation translocations often involve the movement of small numbers of individuals, of typically understudied species, and even data gained from post-release monitoring may remain sparse for a long time before the translocated population establishes and grows. This reduces the suitability of many population models, for which small sample sizes can lead to very high uncertainty, and in turn to incorrect management decisions (Armstrong & Seddon 2008). In **Chapter 2**, I provided a bespoke capture-recapture model for translocated populations (Bickerton *et al.* 2023) that gives more accurate estimates of key population parameters despite limited data. The new model separates the likelihoods for the translocated and non-translocated individuals within the population, constraining the standard JS model to account for the smaller population size and the delay in new entrants joining the population. In the simulation study I illustrated the risk of using non-specific models, showing how the JS model can overestimate population size in the initial establishment phase. This phase is one of the most important for successfully establishing a translocated population: overestimating abundance can bias perception of success and therefore misguide or stop management action when it may be most needed.

5.4 Contributions to the field: Conservation

One of the key drivers for the development of the modified JS model in **Chapter 2** and the use of expert judgement to plan the translocation in **Chapter 4**, was the lack of literature on translocations of prey species, especially reptiles (Bubac *et al.* 2019), into systems where predators are present, especially native predators. This evidence base is further reduced by a publication bias towards successful translocations, as reptile translocations have historically high failure rates (Germano & Bishop 2009). In order to better inform conservation decision making, it is vital to make use of all information available including expert judgement. In **Chapter 4**, I present an example of combining available data with semi-structured interviews with experts to inform a translocation and improve chances of success. Many of the studies the experts had been involved with, and which they referred to in expressing their judgement, were reported in inaccessible grey literature.

The results presented in this thesis reinforce the message that planning monitoring schemes which target clearly defined measures of success is vital for appropriate conservation management. Monitoring should be designed to collect data that can be used to calculate metrics of success, as opposed to many cases where data is collected before deciding on an analytical approach, meaning data may prove irrelevant, inadequate, or even violate key assumptions of models. All the chapters in this thesis focus on species whose cryptic behaviour leads to sparse data, often regardless of monitoring protocols. Where present, this challenge needs to be acknowledged and appropriate methods used or developed, as opposed to using methods that are readily available but not appropriate for the data and as such risk misinforming management decisions (such as the standard JS model in **Chapter 2**). The monitoring and analysis presented in **Chapter 3**, demonstrated the value of regular long-term monitoring schemes in order to establish a baseline understanding of a population. This then allows for detection of population dynamics that could jeopardize the persistence of the species, particularly for vulnerable populations of small size and therefore limited resilience.

5.5 Limitations

There are two main factors that limited this study: the Covid-19 pandemic and the availability of resources and data. Outside of the effects of the pandemic on the progress of the project (see *Covid Impact Statement*), Covid-19 disrupted the standard baseline monitoring carried out by the Mauritian Wildlife Foundation. In the five years prior to the pandemic, and longer in some cases, annual surveys were carried out across all populations mentioned in this thesis. Due to the pandemic and associated restrictions implemented by the Mauritian government, none of the standard surveys were undertaken in 2020. Therefore, when the MV Wakashio oil spill occurred, there was limited recent data to detect impacts and inform urgently needed management decisions. Restrictions also delayed surveys following the oil spill, leading to a 4-month gap between the initial spill and the first surveys; this delay also meant that those first surveys were carried out at the end of the dry season, when resources are most limited and at a different time of year to all previous surveys, preventing accurate comparisons of body condition and health of individuals. Catastrophic events, such as oil spills, extreme weather events and pandemics, are difficult to predict, but their frequency is likely to increase with global change (Meehl *et al.* 2000; Khasnis & Nettleman 2005; Jentsch & Beierkuhnlein 2008). For heavily managed populations of threatened species, contingency plans for rapid response should become standard practice.

Resource limitations and therefore limited data availability are common in conservation. Experimental approaches are often proposed to increase knowledge of a best management strategy, however limited resource availability further reduces statistical power in systems with already small numbers of available individuals. In many cases learning is therefore abandoned in favour of single assumed best strategies (Taylor *et al.* 2017) a pattern accentuated by managers' risk aversion to allocating individuals to treatment groups that may perform worse (e.g. Canessa *et al.* 2020). Conversely, if a single strategy is clearly superior to other strategies, risking an experimental approach may not be optimal. Incorporating learning where needed is key to improving project success as well as informing future projects focusing on the same or other species. To help overcome our Round Island translocation resource constraint we developed an alternative way to justify our single strategy approach, though in future a more formal expert elicitation process would be beneficial.

As previously mentioned, data is likely to be limited when working with cryptic threatened species, so gaining insight into systems can be challenging. A specific example of this is in the dispersal monitoring following the lesser night gecko translocation to Round Island. As found with capture-recapture models, there are few methods to analyse sparse data without high levels of uncertainty (Armstrong & McCarthy 2007), however, collecting greater quantities of data is not always possible. In the case of lesser night geckos, collecting more data would have required more intensive monitoring or the use of more sophisticated tracking technology. The intensity of monitoring was mostly limited by the behavioural ecology of the species which are very small, nocturnal, well camouflaged and inhabit crevices within rocky outcrops that are inaccessible without damaging the habitat.

5.6 Future directions: Statistics

Throughout this thesis, a common theme has been limitation in available data due to: low sample sizes in threatened species; difficulties in monitoring cryptic behaviours; and reduced sampling ability due to a lack of resources. This is likely a common scenario in many species recovery programmes. Therefore, it is vital that the development of bespoke methods for small data continue. The use and availability of these tools, such as the model described in **Chapter 2**, enables decision makers to better understand and predict the outcomes of conservation actions. Bespoke approaches, although technically more challenging, can increase accuracy in parameter estimates and outcomes. This is particularly important when working with multiple stakeholders as increased confidence in potential outcomes of a management action can reassure those that are particularly risk averse (Tulloch *et al.* 2015; Canessa *et al.* 2020). However, the development of bespoke code is more time consuming than using existing R packages and can limit uptake of new approaches, therefore integrating the model described in **Chapter 2** into a more standard framework may be beneficial for future use.

Future investigations could be carried out to investigate the potential impacts of removal on the source population for the Round Island translocation, or impacts of the translocations on other species on Round Island, such as reduced predation pressure on other prey species or changes in population sizes due to increase in prey abundance for predators. These analyses would improve understanding of the interactions between translocated and native species, allowing for an assessment of the translocation at an ecosystem level.

5.7 Future directions: Conservation

Due to the Covid-19 related delay in the lesser night gecko translocation to Round Island, described in **Chapter 4**, only the first seven months of post-release monitoring could be assessed in this thesis. During this time, there were no new entrants to the adult population, only juveniles, therefore use of the modified JS model was not appropriate. Once new adults are observed, use of the modified JS model to estimate population size and survival is recommended. Once the fences are made leaky, then completely open, the model will still be appropriate assuming all individuals are treated as one population and eventually confirm whether this translocation has successfully established (entered a growth then regulation phase; Armstrong & Seddon 2008; IUCN/SSC 2013).

The continued management and monitoring of the reptile populations on islands impacted by the MV Wakashio oil is recommended but unlikely to give rise to further understanding of population dynamics due to the lack of monitoring during the Covid-19 pandemic. However, the analysis of genetic samples collected both before and after the oil spill is close to completion. The combination of this information with the existing knowledge of the system may improve understanding of the long-term impacts of the oil spill and uncover individual level changes that could not be established using population level models.

The translocation example provided in this thesis adds to the limited body of literature on translocations of reptile prey species. On a local scale, the findings can be used to adaptively manage the lesser night gecko population on Round Island, and plan future translocations of other cryptic prey species in Mauritius, such as the Durrell's night gecko (*Nactus durrellorum*) which is currently restricted to Round Island. On a global scale, the initial findings from the Round Island translocation can be used to provide guidance on translocations of species occupying a similar niche. The translocation planning process gives an integrated approach, using literature and expert knowledge to decide on the most suitable release strategy in a scenario with limited information, resources and high uncertainty. Furthermore, analysis of the dispersal data using kernel-based methods and net squared displacement modelling could provide a valuable insight into post-release movements and the efficacy of the soft-release enclosures. The insights from the oil spill study demonstrate the importance of collecting baseline data for populations at with limited resilience to change (e.g. populations with low abundances and species that are slow to mature or reproduce) to enable the detection of deviation in population dynamics. Both of these examples demonstrate the importance of integrating monitoring with management goals, including planning appropriate survey methods to collect the data required to understand if management goals have been met.

To conclude, this thesis provides guidance on the monitoring of threatened cryptic reptile species, following translocations and environmental disasters, to best estimate accurate demographic parameters to enable good management decisions.

References

Aguirre, T.M., Calvo, R., Velasco, M.A., Arellano, M.L., Zarini, O. & Kacoliris, F.P. 2019. Re-establishment of an extinct local population of the Valcheta Frog, *Pleurodema somuncurense*, in a restored habitat in Patagonia, Argentina. *Conservation Evidence* **16**: 48-50.

Akaike, H. 1973. Information theory and the maximum likelihood principle in *2nd International Symposium on Information Theory* (B.N. Petrov and F. Cs ä ki, eds.). Akademiai Ki à do, Budapest.

Akaike, H. 1974. A new look at the statistical model identification. *IEEE Transactions on Automatic Control*, **19(6)**: 716-723.

Albrecht, M.A., Osazuwa-Peters, O.L., Maschinski, J., Bell, T.J., Bowles, M.L., Brumback, W.E., Duquesnel, J., Kunz, M., Lange, J., McCue, K.A., McEachern, A.K., Murray, S., Olwell, P., Pavlovic, N.B., Peterson, C.L., Possley, J., Randall, J.L. & Wright, S.J. Effects of life history and reproduction on recruitment time lags in reintroductions of rare plants. *Conservation Biology*, **33(3)**: 601-611.

Al-Hashem, M. A., Brain, P. F. and Omar, S. A. 2007. Effects of oil pollution at Kuwait's greater Al-Burgan oil field on polycyclic aromatic hydrocarbon concentrations in the tissues of the desert lizard *Acanthodactylus scutellatus* and their ant prey. *Ecotoxicology*, **16(8)**: 551-555.

Allee, W.C. 1931. Animal aggregations: a study in general sociology. Chicago University Press, Chicago.

Armstrong, D.P. & Ewen, J.G. 2001. Assessing the value of follow-up translocations: a case study using New Zealand robins. *Biological Conservation,* **101(2)**: 237-247.

Armstrong, D.P. & McCarthy, M.A. 2007. Big decisions and sparse data: adapting scientific publishing to the needs of practical conservation. *Avian Conservation and Ecology*, **2(2)**: 14.

Armstrong, D. P., Le Coeur, C., Thorne, J. M., Panfylova, J., Lovegrove, T. G., Frost, P. G. & Ewen, J. G. 2017. Using Bayesian mark-recapture modelling to quantify the strength and duration of post-release effects in reintroduced populations. *Biological Conservation,* **215:** 39.

Armstrong, D.P. & Seddon, P.J. 2008. Directions in reintroduction biology. *Trends in Ecology and Evolution*, **23(1)**: 20-25.

Armstrong, D.P. and Wittmer, H.U. 2011. Incorporating Allee effects into reintroduction strategies. *Ecological Research*, **26**: 687-695.

Arnold, E. 1980. Recently extinct reptile populations from Mauritius and Réunion, Indian Ocean. *Journal of Zoology,* **191(1)**: 33-47.

Arnold, E. 2000. Using fossils and phylogenies to understand evolution of reptile communities on islands*. Isolated vertebrate communities in the tropics* (ed. by G. Rheinwald), pp. 309–324. Bonner Zoologische Monographien 46, Museum A. Koenig, Bonn.

Arnold, E. & Jones, C. 1994. The night geckos of the genus *Nactus* in the Mascarene Islands with a description of the distinctive population on Round Island. *Journal of the Wildlife Preservation Trusts,* **30**: 119-131.

Asif, Z., Chen, Z., An, C. & Dong, J. 2022. Environmental impacts and challenges associated with oil spills on shorelines. *Journal of Marine Science and Engineering*, **10(762)**: 1-20.

Bajomi, B., Pullin, A.S., Stewart, G.B. & Takacs-Santa, A. 2010. Bias and dispersal in the animal reintroduction literature. *Oryx*, **44(3)**: 358-365.

Baker, C.M., Bode, M., Dexter, N., Lindenmayer, D.B., Foster, C., MacGregor, C., Plein, M. & McDonald-Madden, E. 2019. A novel approach to assessing the ecosystem-wide impacts of reintroductions. *Ecological Applications*, **29(1)**: e01811.

Balestrieri, A., Gianfranceschi, L., Verduci, F., Gatti, E., Boggioni, P., Mucci, N., Mengoni, C., Capelli, E. and Tremolada, P. 2021. Genetic and viability assessment of a reintroduced Eurasian otter *Lutra lutra* population on the river Ticino, Italy. *Oryx*, **56(4)**: 617-626.

Bam, W., Hooper-Bui, L. M., Strecker, R. M., Adhikari, P. L. and Overton, E. B. 2018. Coupled effects of oil spill and hurricane on saltmarsh terrestrial arthropods. *PloSONE*, **13(4)**: 1-16.

Berger-Tal, O., Blumstein, D.T. & Swaisgood, R.R. 2020. Conservation translocation: a review of common difficulties and promising directions. *Animal Conservation,* **23(2)**: 121-131.

Bertolero, A., Pretus, J.L. & Oro, D. 2018. The importance of including survival release costs when assessing viability in reptile translocations. *Biological Conservation*, **217**: 311-320.

Beyer, J., Trannum, H. C., Bakke, T., Hodson, P. V. and Collier, T. K. 2016. Environmental effects of the Deepwater Horizon oil spill: a review. *Marine Pollution Bulletin*, **110(1)**: 28-51.

Bickerton, K.T., Ewen, J.G., Canessa, S., Cole, N.C., Frost, F., Mootoocurpen, R. & McCrea, R.S. 2023. Avoiding bias in estimates of population size for translocation management. *Ecological Applications*, Accepted Author Manuscript e2918. https://doi-org.chain.kent.ac.uk/10.1002/eap.2918.

Bilby, J. & Moseby, K. 2023. Review of hyperdispersal in wildlife translocations. *Conservation Biology*, e14083.

Bogisch, M., Cree, A., Monks, J.M. 2016. Short-term success of a translocation of Otago skinks (*Oligosoma otagense*) to Orokonui Ecosanctuary. *New Zealand Journal of Zoology*, **43(2)**: 211-220.

Bohm, M., Williams, R., Bramhall, H.R., McMillan, K.M., Davidson, A.D., Garcia, A., Bland, L.M. & Collen, B. 2016. Correlates of extinction risk in squamate reptiles: the relative importance of biology, geography, threat and range size. *Global Ecology and Biogeography*, **25(4)**: 391-405.

Bolam, F.C., Grainger, M.J., Mengersen, K.L., Stewart, G.B., Sutherland, W.J., Runge, M.C. & McGowan, P.J.K. 2019. Using the value of information to improve conservation decision making. *Biological Reviews*, **94**: 629-647.

Bolger, D.T., Morrison, T.A., Vance, B., Lee, D. & Farid, H. 2012. A computer-assisted system for photographic mark-recapture analysis. *Methods in Ecology and Evolution*, **3**: 813-822.

Börger, L., Franconi, N., de Michele, G., Gantz, A., Meschi, F., Manica, A., Lovari, S. & Coulson, T. 2006. Effects of sampling regime on the mean and variance of home range size estimates. *Journal of Animal Ecology*, **75(6)**: 1393-1405.

Börger, L. & Fryxell, J. 2012. Quantifying individual differences in dispersal using net squared displacement. In: Clobert, J., Baguette, M., Benton, T.G. & Bullock, J.M. (Eds.). *Dispersal Ecology and Evolution*. Oxford University Press, Oxford, UK.

Broder, L., Tatin, L., Hochkirch, A., Schuld, A., Pabst L. & Besnard, A. 2020. Optimization of capture-recapture monitoring of illusive species illustrated with a threatened grasshopper. *Conservation Biology*, **34(3)**: 743-753.

Bubac, C.M., Johnson, A.C., Fox, J.A. & Cullingham, C.I. 2019. Conservation translocations and post-release monitoring: identifying trends in failures, biases, and challenges from around the world. *Biological Conservation*, **238**: 1-8.

Buckland, S. T., K. P. Burnham, and N. H. Augustin. 1997. Model selection: An integral part of inference. *Biometrics*, **53**: 603-618.

Bullock, D.J., Arnold, E.N. & Bloxam, Q. 1985. A new endemic gecko (Reptilia: Gekkonidae) from Mauritius. *Journal of Zoology*, **206:** 591-599.

Bullock, D.J. 1986. The ecology and conservation of reptiles on Round Island and Gunner's Quion, Mauritius. *Biological Conservation,* **37(2)**: 135-156.

Bullock, D.J., North, S.G., Dulloo, M.E. & Thorsen, M. 2002. Pages 53-63 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Burns, C.M.B., Olin, J.A., Woltmann, S., Stouffer, P.C. & Taylor, S.S. 2014. Effects of oil of terrestrial verterates: predicting impacts of the Macondo blowout. *BioScience*, **64(9)**:820-828.

Canessa, S., Genta, P., Jesu, R., Lamagni, L., Oneto, F., Salvidio, S. & Ottonello, D. 2016. Challenges of monitoring reintroduction outcomes: insights from the conservation breeding program of an endangered turtle in Italy. *Biological Conservation,* **204**: 128-133.

Canessa, S., Taylor, G., Clarke, R.H., Ingwesen, D., Vandersteen, J. & Ewen, J.G. 2020. Risk aversion and uncertainty create a conundrum for planning recovery of critically endangered species. *Conservation Science and Practice,* **2**: e138.

Carpenter, A.I., Cote, I.M. & Jones, C.G. 2003. Habitat use, egg laying sites and activity patterns of an endangered Mauritian gecko (*Phelsuma guentheri*). *Herpetological Journal,* **13:** 155-157.

Caughley, G. 1994. Directions in conservation biology. *Journal of Animal Ecology*, **63(2)**: 215-244.

Chades, I., McDonnald-Madden, E., McCarthy, M.A., Wintle, B., Linkie M. & Possingham, H.P. 2008. When to stop managing of surveying cryptic threatened species. *Proceedings of the National Academy of Sciences of the United States of America*, **105(37)**: 13936-13940.

Chilvers, B.L., Morgan, K.J. & White, B.J. 2021. Sources and reporting of oil spills and impacts on wildlife 1970-2018. *Environmental Science and Pollution Research*, **28**: 754-762.

Clark, J.D., Huber, D. and Servheen, C. 2002. Bear reintroductions: lessons learnt and challenges. *Ursus*, **13**: 335-345.

Cole, N.C. 2005. 'The ecological impact of the invasive house gecko *Hemidactylus frenatus* upon endemic Mauritian geckos', PhD thesis, University of Bristol, UK.

Cole, N.C., Jones, C.G. & Harris, S. 2005. The need for enemy-free space: the impact of an invasive gecko on island endemics. *Biological Conservation,* **125(4)**: 467-474.

Cole, N., Jones, C., Buckland, S., Mootoocurpen, R., Tatayah, V., Bachraz, V., Nundlaul, V. & Seewajee, P. 2007. *Restoring Island Biodiversity the Reintroduction of Endemic Mauritian Reptile Communities*. Durrell Wildlife Conservation Trust, Mauritian Wildlife Foundation & National Parks and Conservation Service. Jersey, Channel Islands.

Cole, N., Jones, C., Buckland, S., Jhumka, Z., Mootoocurpen, R., Tatayah, V., Bachraz, V., Nundlaul, V., Roopa, P. and Seewajee, P. 2009. *The reintroduction of endangered Mauritian reptiles*. Mauritian Wildlife Foundation. Vacoas, Mauritius.

Cole, N., Goder, M., Roopa, P., Bachraz, V. & Mootoocurpen, R. 2018a. *Leiolopisma telfairii* (errata version published in 2019). The IUCN Red List of Threatened Species 2018: e.T11409A152276731. https://dx.doi.org/10.2305/IUCN.UK.2018-2.RLTS.T11409A152276731.en.

Cole, N., Hector, A., Roopa, P., Mootoocurpen, R. & Goder, M. 2018b. *Casarea dussumieri* (errata version published in 2019). The IUCN Red List of Threatened Species 2018: e.T3989A152276140. https://dx.doi.org/10.2305/IUCN.UK.2018-2.RLTS.T3989A152276140.en.

Cole, N., Mootoocurpen, R. & Nundlaul, V. 2018c. Relative density estimates of Round Island ’s reptiles. *Journal of the Royal Society of Arts and Sciences of Mauritius*, **1:** 1-16.

Cole, N., Mootoocurpen, R., Roopa, P. & Ruhomaun, K. 2021. *Nactus coindemirensis*. The IUCN Red List of Threatened Species 2021: e.T40795A13482515.

Cole, N., Outlaw, L., Vencatasamy, D. & Mootoocurpen, R. 2018d. *Phelsuma guentheri* (errata version published in 2019). The IUCN Red List of Threatened Species 2018: e.T16926A152274946. https://dx.doi.org/10.2305/IUCN.UK.2018-2.RLTS.T16926A152274946.en.

Cole, N. & Payne, C. 2022. *Gongylomorphus bojerii* (amended version of 2015 assessment). The IUCN Red List of Threatened Species 2022: e.T62251A217761064. https://dx.doi.org/10.2305/IUCN.UK.2022-1.RLTS.T62251A217761064.en

Converse, S.J., Moore, C.T. & Armstrong, D.P. 2013. Demographics of reintroduced populations: estimation, modelling and decision analysis. *The Journal of Wildlife Management*, **77(6)**: 1081-1093.

Cormack, R.M. 1964. Estimates of survival from the sighting of marked animals. *Biometrika*, **51:** 429-438.

Couceiro, S.R.M., Hamada, N.H., Ferreira, R.L.M., Forsberg, B.R. and da Silva, J.O. 2007. Domestic sewage and oil spills in streams: effects on edaphic invertebrates in flooded forest, Manaus, Amazonas, Brazil. *Water, Air and Soil Pollution,* **180:**249 – 259.

Courchamp, F., Chapuis, J.L., Pascal, M. 2003. Mammal invaders on islands: impact, control and control impact. *Biological Reviews*, **78**: 347-383.

Courchamp, F., Clutton-Brock, T. and Grenfell, B. 1999. Inverse density dependence and the Allee effect. *Trends in Ecology and Evolution*, **14**: 405-410.

Cox, N., Young, B.E., Bowles, P., Fernandez, M., Marin, J., Rapacciuolo, G., Böhm, M., Brooks, T.M., Hedges, S.B., Hilton-Taylor, C., Hoffmann, M., Jenkins, R.K.B., Tognelli, Alexander, G.J., Allison, A., Ananjeva, N.B., Auliya, M., Avila, L.J., Chapple, D.G., Cisneros-Heredia, D.F., Cogger, H.G., Colli, G.R., de Silva, A., Eisemberg, C.C., Els, J., Fong, A.G., Grant, T.D., Hitchmough, R.A., Iskandar, D.T., Kidera, N., Martins, M., Meiri, S., Mitchell, N.J., Molur, S., Nogueira, C.C., Ortiz, J.C., Penner, J., Rhodin, A.G.J., Rivas, G., Rödel, M.O., Roll, U., Sanders, K.L., Santos-Barrera, G., Shea, G.M., Spawls, S., Stuart, B.L., Tolley, K.A., Trape, J.F., Vidal, M.A., Wagner, P., Wallace, B.P., Xie, Y. 2022. A global reptile assessment highlights shared conservation needs of tetrapods. *Nature*, **605**: 285-290.

Crall, J. P., Stewart, C. V., Berger-Wolf, T. Y., Rubenstein, D. I. and Sundaresan, S. R. 2013. *HotSpotter — Patterned species instance recognition*. 2013 IEEE Workshop on Applications of Computer Vision (WACV): 230-237. DOI: 10.1109/WACV.2013.6475023.

Crimmins, S.M., Roberts, N.M., Hamilton, D.A. and Mynsberge, A.R. 2009. Seasonal detection rates of otters (*Lontra canadensis*) using bridge-site and random-site surveys. *Canadian Journal of Zoology*, **87(11)**: 993-999.

DATLife Database. 2021. Max-Planck Institute for Demographic Research (Germany). Available at www.datlife.org (data downloaded on 08 August 2021).

Deredec, A. & Courchamp, F. 2007. Importance of the Allee effect for reintroductions. *Ecoscience*, **14(4)**: 440-451.

DiCiccio, T.J. and Efron, B. 1996. Bootstrap confidence intervals. *Statistical Science*, **11(3)**: 189-288.

Dickens, M.J., Delehanty, D.J. & Romero, L.M. 2010. Stress: an inevitable component of animal translocation. *Biological Conservation*, **143(6)**: 1329-1341.

Dieterman, J.D., Painovich, J.F., Painovich, N. & Staples, D.F. 2010. Lake sturgeon population status and demography in the Kettle River, Minnesota, 1992-2007. *North American Journal of Fisheries Management,* **30**: 337-351.

Dimond, W.J. & Armstrong, D.P. 2007. Adaptive harvesting of source populations for translocation: a case study with New Zealand robins. *Conservation Biology,* **21(1)**: 114-124.

Dolny, A., Sigutova, H., Ozana, S. & Choleva L. 2018. How difficult is it to reintroduce a dragonfly? Fifteen years monitoring *Leucorrhinia dubia* at the receiving site. *Biological Conservation*, **218**: 110-117.

Ducatez, S. & Shine, R. 2019. Life-history traits and the fate of translocated populations. *Conservation Biology*, **33(4)**: 853-860.

Earnhardt, J., Velez-Valentin, J., Valentin, R., Long, S., Lynch, C. & Schowe, K. 2014. The Puerto Rican parrot reintroduction program: sustainable management of the aviary population. *Zoo Biology*, **33(2)**: 89-98.

Emery, J.P., Valentine, L.E., Hitchen, Y. & Mitchell, N. 2021. Survival of an Extinct in the Wild skink from Christmas Island is reduced by an invasive centipede: implications for future reintroductions. *Biological Invasions*, **23(2)**: 581-592.

Esler, D., Ballachey, B.E., Matkin, C., Cushing, D., Kaler, R., Bodkin, J., Monson, D., Esslinger, G. & Kloecker, K. 2018. Timelines and mechanisms of wildlife population recovery following the Exxon Valdez oil spill. *Deep-Sea Research Part II: Topical Studies in Oceanography,* **147:** 36-42.

Evans, M.J., Pierson, J.C., Neaves, L.E., Gordon, I.J., Ross, C.E., Brockett, B., Rapley, S., Wilson, B.A., Smith, K.J., Andrewartha, T., Humphries, N. & Manning, A.D. 2023. Trends in animal translocation research. *Ecography*, **2023(3):** 1-20.

Ewen, J.G. & Armstrong, D.P. 2008. Strategic monitoring of reintroductions in ecological restoration programmes. *Ecoscience*, **14(4)**: 401-409.

Ewen, J.G., Soorae, P. & Canessa, S. 2014. Reintroduction objectives, decisions and outcomes: Global perspectives from the herpetofauna. *Animal Conservation*, **17**: 74-81.

Falk, B.G., Snow, R.W. & Reed, R.N. 2017. A validation of 11 body-condition indices in a giant snake species that exhibits positive allometry. *PLoS ONE*, **12(7)**: 1-20.

Farley, S.S., Dawson, A., Goring, S.J. & Williams, J.W. 2018. Situating ecology as a big-data science: Current advances, challenges, and solutions. *BioScience*, **68(8)**: 563-576.

Ferner, J.W. & Plummer, M.V. 2016. Marking and measuring reptiles. In: Dodd, C.K. (Eds.). *Reptile Ecology and Conservation: A handbook of techniques*. Oxford University Press, Oxford, UK.

Fernandes, G.M., Martins, D.A., dos Santos, R.P., de Santiago, I. 2022. Levels, source appointment, and ecological risk of petroleum hydrocarbons in tropical coastal ecosystems (northeast Brazil): Baseline for future monitoring programmes of an oil spill area. *Environmental Pollution*, **296**: 1-27.

Ferretti, F., Corazza, M., Campana, I., Pietrocini, V., Brunetti, C., Scornavacca, D. & Lovari, S. 2015. Competition between wild herbivores: reintroduced red deer and Apennine chamois. *Behavioural Ecology*, **26(2)**: 550-559.

Fischer, J. & Lindenmayer, D.B. 2000. An assessment of the published results of animal relocations. *Biological Conservation*, **96**: 1-11.

Flynn-Plummer, T.P. & Monks, J.M. 2021. Penned release reduces area use by translocated barking geckos (*Naultinus punctatus*). *New Zealand Journal of Ecology*, **45(1):** 1-7.

Frankel, O.H. 1970. Variation – the essence of life. *Proceedings of the Linnean Society of New South Wales*, **95(2)**: 158-169.

Frankham, R. 2005. Genetics and extinction. *Biological Conservation*, **126(2)**: 131-140.

Fritts, T. H. and McGehee, M. A. 1982. *Effects of petroleum on the development and survival of marine turtle embryos*. US Department of the Interior/Minerals Management Service, Gulf of Mexico.

Germano, J.M. & Bishop, P.J. 2009. Suitability of amphibians and reptiles for translocation. *Conservation Biology*, **23(1):** 7-15.

Gimenez, O., Lebreton, J.D., Choquet, R. & Pradel, R. 2018. R2ucare: An R package to perform goodness-of-fit tests for capture–recapture models. *Methods in Ecology and Evolution*, **9(7)**, 1749-1754. https://oliviergimenez.github.io/R2ucare/.

Goble, C. & Goetz, M. 2011. *Husbandry guidelines for the Mauritian lesser night gecko (Nactus coindemirensis)*. Jersey, Channel Islands, UK: Durrell Wildlife Conservation Trust.

Griffith, B., Scott, J.M., Carpenter, J.W. and Reed, C. 1989. Translocation as a species conservation tool: status and strategy. *Science*, **245**: 477-480.

Griffiths, C.J., Jones, C.G., Hansen, D.M., Puttoo, M., Tatayah, R.V., Muller, C.B. & Harris, S. 2010. The use of extant non-indigenous tortoises as a restoration tool to replace extinct ecosystem engineers. *Restoration Ecology,* **18(1):** 1-7.

Gurumoorthi, K., Suneel, V., Rao, V.T., Thomas, A.P. & Alex, M.J. 2021. Fate of MV Wakashio oil spill off Mauritius coast through modelling and remote sensing observations. *Marine Pollution Bulletin*, **173**: 112892.

Hampton, S.E., Strasser, C.A., Tewksbury, J.J., Fram, W.K., Budden, A.E., Batcheller, A.L., Duke, C.S. & Porter, J.H. 2013. Big data and the future of ecology. *Frontiers in Ecology and the Environment*, **11(3)**: 156-162.

Hernandez, P.A., Graham, C.H., Master, L.L. & Albert, D.L. 2006. The effect of sample size and species characteristics on performance of different species distribution modelling methods. *Ecography*, **29**: 773-785.

Hurvich, C.M. and Tsai, C.L. 1989. Regression and time series model selection in small samples. *Biometrika,* **76**: 297-307.

IPBES. 2019. *Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. S. Díaz, J. Settele, E. S. Brondízio E.S., H. T. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, K. A. Brauman, S. H. M. Butchart, K. M. A. Chan, L. A. Garibaldi, K. Ichii, J. Liu, S. M. Subramanian, G. F. Midgley, P. Miloslavich, Z. Molnár, D. Obura, A. Pfaff, S. Polasky, A. Purvis, J. Razzaque, B. Reyers, R. Roy Chowdhury, Y. J. Shin, I. J. Visseren-Hamakers, K. J. Willis, and C. N. Zayas (eds.). IPBES secretariat, Bonn, Germany. 56pages.

IUCN. 2022. The IUCN Red List of Threatened Species. Version 2022-2. https://www.iucnredlist.org. Accessed on [02 Aug 2023].

IUCN/SSC. 2013. *Guidelines for Reintroductions and Other Conservation Translocations*. Gland, Switzerland: IUCN Species Survival Commission.

Jackson, C.H.N. 1933. On the true density of tsetse flies. *Journal of Animal Ecology*, **2**: 204-209.

Jackson, C.H.N. 1939. The analysis of an animal population. *Journal of Animal Ecology,* **8**: 236-246.

Jensen, M.A., Paton, D.C. & Moseby, K.E. 2021. Delayed release improves site fidelity but has little effect on survival or breeding success of reintroduced western quolls (*Dasyurus geoffroii*). *Austral Ecology*, **46**: 1298-1310.

Jentsch, A. & Beierkuhnlein, C. 2008. Research frontiers in climate change: effects of extreme meteorological events on ecosystems. *Comptes Rendus Geoscience*, **240(9-10)**: 621-628.

Jolly, G.M. 1965. Explicit estimates from capture-recapture data with both death and immigration-stochastic model. *Biometrika*, **52**: 225-247.

Jones, C.G. 1993. Ecology and conservation of Mauritian skinks. *Proceedings of the Royal Society of Arts and Sciences of Mauritius,* **5(3)**: 71-95.

Khasnis, A.A. & Nettleman, M.D. 2005. Global warming and infectious disease. *Archives of Medical Research*, **36(6)**: 689-696.

King, R. and McCrea, R.S. Capture-recapture methods and models: Estimating population size. In *Handbook of Statistics: Integrated Population Biology and Modeling, Part B, Volume 40*, edited by Srinivasa Rao, A.S.R. & Rao, C.R, 33-83. Amsterdam: Elsevier, 2019.

Knox, C.D. & Monks, J.M. 2014. Penning prior to release decreases post-translocation dispersal of jewelled geckos. *Animal Conservation*, **17(1):** 18-26.

Knox, C.D., Jarvie, S., Easton, L.J. & Monks, J.M. 2017. Soft-release, but not cool winter temperatures, reduced post-translocation dispersal of jewelled geckos. *Journal of Herpetology*, **51(4)**: 490-496.

Laake, J.L. 2013. RMark: An R Interface for Analysis of Capture-Recapture Data with MARK. AFSC Processed Rep, 1:25. Alaska Fisheries Science Center, NOAA, Seattle, USA.

Laake, J.L., Johnson, D.S. & Conn, P.B. 2013. "marked: An R package for maximum-likelihood and MCMC analysis of capture-recapture data." Methods in Ecology and Evolution 4: 885-890.

Lebreton, J., Burnham, K.P., Clobert, J. & Anderson, D.R. 1992. Modeling survival and testing biological hypotheses using marked animals: a unified approach with case studies. *Ecological Monographs*, **62(1)**: 67-118.

Lettink, M., Norbury, G., Cree, A., Seddon, P.J., Duncan, R.P. & Schwarz, C.J. 2010. Removal of introduced predators, but not artificial refuge supplementation, increases skink survival in coastal duneland. *Biological Conservation*, **143**: 72-77.

Lewis, D. 2020. How Mauritius is cleaning up after major oil spill in biodiversity hotspot. *Nature*, **585:** 172.

Linhoff, L.J. & Donnelly, M.A. 2022. Assessing release strategies for reintroductions of endangered Wyoming toads. *Wildlife Society Bulletin,* **46(5)**: 1-11.

Malcolm HM, Shore RF. 2003. Effects of PAHs on terrestrial and freshwater birds, mammals, and amphibians. Pages 225–242 in Douben PE, ed. *PAHs: An Ecotoxicological Perspective*. Wiley.

Martin, T.G., Burgman, M.A., Fidler, F., Kuhnert, P.M., Low-Choy, S., McBride, M. & Mengersen, K. 2012. Eliciting expert knowledge in conservation science. *Conservation Biology*, **26(1)**: 29-38.

Matechou, E., McCrea, R.S., Morgan, B.J., Nash, D.J. & Griffiths, R.A. 2016. Open models for removal data. *The Annals of Applied Science*, **10(3)**: 1572-1589.

MATLAB. 2023. MATLAB (version 9.14 (R2023a)). Natick, Massachusetts: The Mathworks Inc.

McCleery, R., Guralnick, R., Beatty, M., Belitz, M., Campbell, C.J., Idec, J., Jones, M., Kang, Y., Potash, A. & Fletcher Jr, R.J. 2023. Uniting Experiments and Big Data to advance ecology and conservation. *Trends in Ecology and Evolution*, **38(10)**: 970-979.

McCrea, R., King, R., Graham, L. & Börger, L. 2023. Realising the promise of large data and complex models. *Methods in Ecology and Evolution*, **14**: 4-11.

Meehl, G.A., Zwiers, F., Evans, J., Knutson, T., Mearns, L. & Whetton, P. 2000. Trends in extreme weather and climate events: issues related to modelling extremes in projections of future climate change. *Bulletin of the American Meteorological Society*, **81(3)**: 427-436.

Michaelides, S., Cole, N. & Funk, S.M. 2015. Translocation retains genetic diversity of a threatened endemic reptile in Mauritius. *Conservation Genetics*, **16(3)**: 661-672.

de Milliano, J., Stefano, J.D., Courtney, P., Temple-Smith, P. & Coulson, G. 2016. Soft-release versus hard-release for reintroduction of an endangered species: An experimental comparison using eastern barred bandicoots (Perameles gunnii). *Wildlife Research*, **43**: 1-12.

Morris, S.D., Brook, B.W., Moseby, K.E. & Johnson, C.N. 2021. Factors affecting success of conservation translocations of terrestrial vertebrates: A global systematic review. *Global Ecology and Conservation*, **28**: 1-15.

Moseby, K.E., Hill, B.M. and Lavery, T.H. 2014. Tailoring release protocols to individual species and sites: one size does not fit all. *PLOS One*, **9(6)**: 1-12.

Moseby, K.E., Lollback, G.W. & Lynch C.E. 2018. Too much of a good thing; successful reintroduction leads to overpopulation in a threatened mammal. *Biological Conservation*, **219**: 78-88.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B. & Kent, J. 2000. Biodiversity hotspots for conservation priorities. *Nature*, **403**: 853-858.

Nafus, M.G., Esque, T.C., Averill-Murray, R.C., Nussear, K.E. & Swaisgood, R.R. 2017. Habitat drives dispersal and survival of translocated juvenile desert tortoises. *Journal of Applied Ecology,* **54:** 430-438.

Niu, S., Wang, S., Wang, J., Xia, J. & Yu, G. 2020. Integrative ecology in the era of big data - From observation to prediction. *Science China Earth Sciences*, **63(10)**: 1429-1442.

Ovaskainen, O. & Meerson, B. 2010. Stochastic models of population extinction. *Trends in Ecology and Evolution*, **25(11)**: 643-652.

Panfylova, J., Ewen, J.G. & Armstrong, D.P. 2019. Making structured decisions for reintroduced populations in the face of uncertainty. *Conservation Science and Practice*, **1(10)**: 1-9.

Peig, J. & Green A.J. 2009. New perspectives for estimating body condition from mass/length data: the scaled mass index as an alternative method. *Oikos*, **118:** 1883-1891.

Peig, J. & Green A.J. 2010. The paradigm of body condition: A critical reappraisal of current methods based on mass and length. *Functional Ecology*, **24(6):** 1323-1332.

Pernetta, A.P., Bell, D.J. & Jones, C.G. 2005. Macro- and microhabitat use of Telfair's skink (*Leiolopisma telfairii*) on Round Island, Mauritius: implications for their translocation. *Acta Oecologica*, **28(3)**: 313-323.

du Plessis, S. J., Howard-McCombe, J., Melvin, Z. E., Sheppard, E. C., Russo, I. R. M., Mootoocurpen, R., & Bruford, M. W. 2019. Genetic diversity and cryptic population re-establishment: Management implications for the Bojer's skink (*Gongylomorphus bojerii*). *Conservation Genetics*, **20**: 137-152.

Pollock, K.H., Nichols, J.D., Brownie, C. & Hines, J.E. 1990. Statistical inference for capture-recapture experiments. *Wildlife Monographs*, **107**: 3-97.

Pollock, K.H. 2000. Capture-recapture models. *Journal of the American Statistical Association*, **95(449)**: 293-296.

Pradel, R. 1996. Utilization of capture-mark-recapture for the study of recruitment and population growth. *Biometrics*, **52**: 703-709.

R Core Team. 2023. R: A language and environment for statistical computing (version 4.3.0). Vienna, Austria: R Foundation for Statistical Computing. Available at: https://www.R-project.org/

Rasmussen A.R., Murphy J.C., Ompi M., Gibbons J.W. & Uetz P. 2011. Marine Reptiles. *PLoS ONE*, **6(11)**: e27373. doi:10.1371/journal.pone.0027373

Resende, P.S., Viana-Junior, A.B., Young, R.J. and Azevedo, C.S. 2021. What is better for animal conservation translocation programmes: Soft- or hard-release? A phylogenetic meta-analytical approach. *Journal of Applied Ecology*, **58**: 1122-1132.

Romijn, R.L. & Hartley, S. 2016. Trends in lizard translocations in New Zealand between 1988 and 2013. *New Zealand Journal of Zoology*, **43(2)**: 191-210.

Round Island Management Plan 2019-2025. Mauritian Wildlife Foundation, National Parks and Conservation Service, Forestry Service, Durrell Wildlife Conservation Trust. Vacoas, Mauritius.

Runge, M.C., Converse, S.J. & Lyons, J.E. 2011. Which uncertainty? Using expert elicitation and expected value of information to design an adaptive program. *Biological Conservation*, **144**: 1214-1223.

Salguero-Gomez, R., Jones, O.R., Jongejans, E., Blomberg, S.P., Hodgson, D.J., Mbeau-Ache, C., Zuidema, P.A., de Kroon, H. & Buckley, Y. 2016. Fast-slow continuum and reproductive strategies structure plant life-history variation worldwide. *PNAS*, **113(1)**: 230-235.

Samaranayaka, A. & Fletcher, D. 2010. Modelling environmental stochasticity in adult survival for a long-lived species. *Ecological Modelling*, **221(3):** 423-427.

Scarlett, A.G., Nelson, R.K., Gagnon, M.M., Holman, A.I., Reddy, C.M., Sutton, P.A & Grice, K. 2021. MV Wakashio grounding incident in Mauritius 2020: The world's first major spillage of Very Low Sulfur Fuel Oil. *Marine Pollution Bulletin*, **171**: 1-11.

Schwarz, C.J. & Arnason, A.N. 1996. A general methodology for the analysis of capture-recapture experiments in open populations. *Biometrics*, **52(3)**: 860-873.

Seaman, D.E., Millspaugh, J.J., Kernohan, B.J., Brundige, G.C., Raedeke, K.J. & Gitzen, R.A. 1999. Effects of sample size on kernel home range estimates. *Journal of Wildlife Management*, **63(2)**: 739-747.

Seber, G.A.F. 1965. A note on the multiple recapture census. *Biometrika*, **52**: 249-259.

Seber, G.A.F. 1982. *The estimation of animal abundance*. Edward Arnold, London, UK.

Seber, G.A.F. 1986. A review of estimating animal abundance. *Biometrics*, **42**: 267-292.

Seddon, P.J., Soorae, P.S. & Launay, F. 2005. Taxonomic bias in reintroduction projects. *Animal Conservation*, **8**: 51-58.

Seddon, P.J., Armstrong, D.P. & Maloney, R.F. 2007. Developing the science of reintroduction biology. *Conservation Biology*, **21(2)**: 303-312.

Seddon, P.J. 2010. From reintroduction to assisted colonization: moving along the conservation translocation spectrum. *Restoration Ecology*, **18(6)**: 796-802.

Seddon, P. and Armstrong, D. 2016. 2. Reintroduction and Other Conservation Translocations: History and Future Developments. In: Jachowski, D., Millspaugh, J., Angermeier, P. and Slotow, R. ed. *Reintroduction of Fish and Wildlife Populations*. Berkeley: University of California Press, pp. 7-28.

Seddon, P.J., Strauss, W.M. & Innes, J. 2012. Animal translocations: What are they and why do we do them? In: Ewen, J.G., Armstrong, D.P. & Parker, K.A. (Eds.), *Reintroduction biology: integrating science and management*. Wiley-Blackwell, West Sussex, UK.

Seddon, P.J., Griffiths, C.J., Soorae, P.S. and Armstrong, D.P. 2014. Reversing defaunation: restoring species in a changing world. *Science*, **345(6192)**: 406-412.

Short JW, Irvine GV, Mann DH, Maselko JM, Pella JJ, Lindeberg MR, Payne JR, Driskell WB, Rice SD. 2007. Slightly weathered Exxon Valdez oil persists in Gulf of Alaska beach sediments after 16 years. *Environmental Science and Technology,* **41:** 1245–1250.

Sinclair, A.R.E., Pech, R.P., Dickman, C.R., Hik, D., Mahon, P. & Newsome, A.E. 1998. Predicting effects of predation on conservation of endangered prey. *Conservation Biology*, **12(3)**: 564-575.

Smith, L.L., Steen, D.A., Conner, L.M. & Rutledge, J.C. 2013. Effects of predator exclusion on nest and hatchling survival in the gopher tortoise. *Journal of Wildlife Management*, **77(2)**: 352-358.

Soorae, P. (ed.) 2008. *Global re-introduction perspectives: re-introduction case-studies from around the globe*. IUCN/SSC Re-introduction Specialist Group.

Soorae, P. (ed.) 2010. *Global re-introduction perspectives: 2010. Additional case-studies from around the globe*. IUCN/SSC Re-introduction Specialist Group, IUCN/SSC Re-introduction Specialist Group and Environment Agency.

Soorae, P. (ed.) 2011. *Global re-introduction perspectives: 2011. More case studies from around the globe*. IUCN/SSC Re-introduction Specialist Group and Environment Agency.

Soorae, P. (ed.) 2013. *Global re-introduction perspectives: 2013. Further case studies from around the globe*. IUCN/SSC Re-introduction Specialist Group and Environment Agency.

Soorae, P. (ed.) 2016. *Global re-introduction perspectives: 2016. Case-studies from around the globe.* IUCN/SSC Reintroduction Specialist Group and Environment Agency.

Soorae, P. (ed.) 2018. *Global reintroduction perspectives: 2018. Case studies from around the globe.* IUCN/SSC Reintroduction Specialist Group and Environment Agency.

Soorae, P. (ed.) 2021. *Global conservation translocation perspectives: 2021. Case studies from around the globe.* IUCN SSC Conservation Translocation Specialist Group, Environment Agency and Calgary Zoo.

Southwell, D., Legge, S., Woinarski, J., Lindenmayer, D., Lavery, T. & Wintle, B. 2021. Design considerations for rapid biodiversity reconnaissance surveys and long-term monitoring to assess the impact of wildfire. *Diversity and Distribution*, **28(3)**: 559-570.

Stoddart, D. R., Peake, J.F., Gordon, C., & Burleigh, R. 1979. Historical records of Indian Ocean giant tortoise populations. *Philosophical Transactions of the Royal Society of London Series B,* **268(1011)**: 147–158.

Stokeld, D., Fisher, A., Gentles, T., Hill, B.M., Woinarski, J.C.Z., Young, S. & Gillespie, G.R. 2018. Rapid increase of Australian tropical savanna reptile abundance following exclusion of feral cats. *Biological Conservation*, **225**: 213-221.

Sutherland, W.J., Armstrong, D., Butchart, S.H.M., Earnhardt, J.M., Ewen, J.G., Jamieson, I., Jones, C.G., Lee, R., Newbery, P., Nichols, D., Parker, K.A., Sarrazin, F., Seddon, P.J., Shah, N. and Tatayah, V. 2010. Standards for documenting and monitoring bird reintroduction projects. *Conservation Letters,* **3(4):** 229-235.

Swan, K.D., Lloyd, N.A. & Moehrenschlager, A. 2018. Projecting further increases in conservation translocations: a Canadian case study. *Biological Conservation*, **228**: 175-182.

Tanentzap, A.J. & Lloyd, K.M. 2017. Fencing in nature? Predator exclusion restores habitat for native fauna and leads biodiversity to spill over into the wider landscape. *Biological Conservation,* **214**: 119-126.

Taylor, G., Canessa, S., Clarke, R.H., Ingwersen, D., Armstrong, D.P., Seddon, P.J. & Ewen, J.G. 2017. Is reintroduction biology an effective applied science? *Trends in Ecology and Evolution,* **32(11)**: 873-880.

Teixeira, C.P., de Azevedo, C.S., Mendl, M., Cipreste, C.F. & Young, R.J. 2007. Revisiting translocation and reintroduction programmes: the importance of considering stress. *Animal Behaviour*, **73**: 1-13.

Tetzlaff, S.J., Sperry, J.H. & DeGregorio, B.A. 2019. Effects of anitpredator training, environmental enrichment, and soft release on wildlife translocation: a review and meta-analysis. *Biological Conservation*, **236**: 324-331.

Tobajas, J., Descalzo, E., Villafuerte, R., Jiminez, J., Mateo, R. & Ferreras, P. 2020. Conditioned odor aversion as a tool for reducing post-release predation during animal translocations. *Animal Conservation*, **24(3)**: 373-385.

Todman, L.C., Bush, A. & Hood, A.S.C. 2023. 'Small data' for big insights in ecology. *Trends in Ecology and Evolution*, **38(7)**: 615-622.

Towns, D.R., Hitchmough, R.A., Perrott, J. 2016. Conservation of New Zealand Lizards: A Fauna Not Forgotten but Undervalued?. In: Chapple, D. (eds) *New Zealand Lizards*. Springer, Cham.

Tuberville, T.D., Clark, E.E., Buhlmann, K.A. & Gibbons, J.W. 2005. Translocation as a conservation tool: site fidelity and movement of repatriated gopher tortoises (*Gopherus polyphemus*). *Animal Conservation*, **8**: 349-358.

Turko, A.J., Leclair, A.T.A, Mandrak, N.E., Drake, D.A.R., Scott, G.R. and Pitcher T.E. 2021. Choosing source populations for conservation reintroductions: lessons from variation in thermal tolerance among populations of the imperilled redside dace. *Canadian Journal of Fisheries and Aquatic Sciences*, **78(9)**: 1347-1355.

Urian, K., Gorgone, A., Read, A., Balmer, B., Wells, R.S., Berggren, P., Durban, J., Eguchi, T., Rayment, W. & Hammond, P.S. 2015. Recommendations for photo-identification methods used in capture-recapture models with cetaceans. *Marine Mammal Science*, **31(1)**: 298-321.

White, G.C. & K. P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. *Bird Study,* **46**, 120-138.

Wikelski, M., Wong, V., Chevalier, B., Rattenborg, N. and Snell, H. L. 2002. Marine iguanas die from trace oil pollution. *Nature*, **417(6889):** 607-608.

Zhou, M., McCrea, R.S., Matechou, E., Cole, D.J. & Griffiths, R.A. 2019. Removal models accounting for temporary emigration. *Biometrics*, **75**: 24-35.

Zychowski, G.V. & Godard-Codding, C.A.J. 2017. Reptilian exposure to polycyclic aromatic hydrocarbons and associated effects. *Environmental Toxicology and Chemistry*, **36(1)**: 25-35.