NEW ZEALAND JOURNAL OF ECOLOGY

RESEARCH

Eradication confirmation of mice from Antipodes Island and subsequent terrestrial bird recovery

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Published online: 28 November 2022

Abstract: Antipodes Island is part of New Zealand's World Heritage subantarctic region and hosts special biodiversity values and significant species endemism. Invasive house mice were the only introduced mammal and detrimentally impacted invertebrate and native bird communities. Eradication of mice from Antipodes Island was undertaken in 2016 and confirmed in 2018. We present the monitoring used to confirm eradication of mice and the ecological outcomes measured over the 6 years since the eradication. Result monitoring for confirmation applied a simple regime to search for mice following a delay of two mouse breeding seasons since baiting was completed. Outcome monitoring targeted endemic land bird taxa for possible changes due to operational impacts and ecological recovery following eradication of mice. The operation had no long-term negative impacts and endemic land bird taxa have recovered quickly from variable levels of non-target mortality. Estimates of abundance of Antipodes Island snipe, Antipodes Island pipit and Reischek's parakeet showed strong long-term positive response to mouse eradication.

Keywords: Antipodes Island, eradication, house mouse, Million Dollar Mouse, *Mus musculus*, subantarctic

Introduction

Islands are biodiversity hotspots and priority targets for conservation actions globally (Jones et al. 2016; Holmes et al. 2019). The global extinction crisis is exacerbated on islands, which are highly vulnerable to ecosystem degradation and extinctions from invasive vertebrate pests (Angel et al. 2009; Tershy et al. 2015; Jones et al. 2016; Holmes et al. 2019). Invasive rodents, particularly *Rattus* spp., are some of the most widely spread invasive mammals on the planet and extremely detrimental to biodiversity values on islands (Towns et al. 2006). More recently, invasive house mice (*Mus musculus*) have also been recognized as a similar and significant threat, with their omnivorous diet causing wide-ranging ecosystem impacts on islands (Angel et al. 2009; Eriksson et al. 2014; Broome et al. 2019; Duhr et al. 2019; Murphy & Nathan 2021). Mice can have severe impacts on their prey on islands including dramatic reductions in invertebrate, lizard and bird abundances (Murphy & Nathan 2021). On subantarctic islands mice have had severe impacts on seabird populations through predation of eggs, chicks and adults (Wanless et al. 2012; Davies et al. 2015; Dilley et al. 2015, 2016; Russell et al. 2020a). In the New Zealand subantarctic region, mice established in the absence of rats on Auckland Island, Enderby Island and Antipodes Island (Russell et al. 2022).

Mice probably arrived on Antipodes Island in 1908 with the shipwreck of the French barque 'President Felix Faure' and are genetically distinct from other New Zealand populations (Veale et al. 2018). Since arrival, mice have been the only mammalian pest species present and inhabited all parts of the main island but have never been detected on other islands in the group (Russell 2012).

Mice were widespread on Antipodes Island and population density in winter was high, between 74 and 104 mice ha^{-1} (Elliott et al. 2015). Comparative studies between Antipodes Island and analogous habitat on the nearby pest-free islands reveal the extensive ecological damage caused by mice. They have been particularly damaging to invertebrates, affecting their abundance, composition and distribution (Russell 2012). Impacts have been especially prevalent for preferred prey species such as medium-sized flightless invertebrates, causing extirpation of at least three taxa (Marris 2000; McIntosh 2001; Russell et al. 2020b). Damage to birds has been less described. Distributed extensively across Antipodes Island are two endemic land bird species: Antipodes Island parakeet (*Cyanoramphus unicolor*) and Reischek's parakeet (*Cyanoramphus hochstetteri*); and two endemic sub-species of land birds, Antipodes Island snipe (*Coenocorypha aucklandica meinertzhagenae*) and Antipodes Island pipit (*Anthus novaeseelandiae steindachneri*). Mice compete with

DOI: https://doi.org/10.20417/nzjecol.46.3488

the land birds for food (Imber et al. 2005). Unusually, snipe on Antipodes Island exhibit bi-modal breeding; and have historically had much lower abundance than on other New Zealand subantarctic islands, probably due to mice (Miskelly et al. 2006). As well as the land birds, the Antipodes Islands are a breeding ground for 21 seabird species (Tennyson et al. 2002) including the endemic Antipodean wandering albatross (*Diomedea antipodensis*), New Zealand's largest population of grey petrels (*Procellaria cinerea*), colonies of erect-crested penguins (*Eudyptes sclateri*), eastern rockhopper penguins (*Eudyptes filholi moseseyi*); and seals – the New Zealand fur seal (*Arctocephalus forsteri*), subantarctic fur seal (*Arctocephalus tropicalis*) and southern elephant seal (*Mirounga leonina*). At least three of the ten burrowing seabird species: black-bellied storm petrel (*Fregetta tropica*), grey-backed storm petrel (*Garrodia nereis*) and subantarctic little shearwater (*Puffinus elegans*) appear to have been supressed by mice (Imber et al. 2005).

Eradication is a high impact conservation management tool and increasingly important for recovery and retention of native and endemic biodiversity on islands (Howald et al. 2007; Jones et al. 2016; Brooke et al. 2017; Holmes et al. 2019; Spatz et al. 2022). In recent years, capability and confidence in eradicating mice has advanced with greater than 90% of projects applying New Zealand's current best practice on temperate islands achieving success (Broome et al. 2019). In winter 2016, a mouse eradication operation was implemented on Antipodes Island using helicopters to spread rodent bait (Pestoff 20R®) in two applications (16 kg ha^{-1} and 8 kg ha^{-1} respectively; Horn et al. 2019). Prior to the eradication, between 2014–2016, a programme of preparation work was undertaken to support the team and protect helicopters: field hut renovations, temporary hangar construction, clearing helipads for refuelling and baiting operations, and installation of one shelter at the bait loading site and two near the field hut. Landslips occurred in January 2014, damaging the field hut and temporarily reducing the vegetated area of Antipodes Island by approximately 15% (GPE and KJW pers. obs. 2014). In 2016, installation of the temporary helicopter platform and hangar required clearing vegetation and levelling a site approximately 32 m × 11 m within an area where vegetation had slipped in 2014. All temporary infrastructure was removed 2 months later, on completion of the baiting operation (Horn et al. 2019).

The response of native species to eradication is generally poorly monitored and under-reported (Jones et al. 2016; Segel et al. 2021). Ecological monitoring on Antipodes Island is constrained by poor access because of challenging logistics due to its remoteness, steep topography and the high cost of transport. Allowing for the limitations, endemic land bird taxa were identified as key indicators to efficiently monitor the impacts of eradication activities on non-target animals and measure ecosystem response over time (Elliott et al. 2015). In this paper we present: (1) the monitoring effort used to confirm the result of the mouse eradication attempt; (2) the response of endemic land bird taxa 6 years after mice were eradicated; and (3) anecdotal observations of other ecosystem changes.

Methods

Site description

The Antipodes Islands (2100 ha) are located in the Southern Ocean at 49.69°S, 178.77°E, 733 km southeast of Dunedin, New Zealand (Fig. 1). The islands are uninhabited and include the main Antipodes Island and six smaller islands. They are a Nature Reserve and one of five island groups that comprise the New Zealand subantarctic region listed as a UNESCO World Heritage Site in 1998 for outstanding biodiversity values including a high level of endemism (Russell et al. 2022). New Zealand's Department of Conservation (DOC) administers the site and visits are by permit only. The coastline is generally inaccessible being cliff-bound and having no harbour. There are two landing points for small boats near Anchorage Bay in the north but landing is only possible in calm conditions.

The island has two broad ecological zones – coastal and inland (Elliott et al. 2015). The coast is dominated by rock faces and dense *Poa litorosa* tussock grasslands up to 1.5 m high, also containing *Carex trifida*, *C. appressa*, *Poa foliosa* and coastal turfs. From the steep coast the island rises to a plateau of shorter grasslands with deep peat soils still dominated by *Poa litorosa* associated with prickly shield fern (*Polystichum vestitum*), ferns, herbs and low shrubs (*Coprosma rugosa*) (Godley 1989). Areas of peat bog are dominated by sedges and megaherb species. Williams et al. (2007) identified three rare ecosystems of high fertility – seabird guano deposits, seabird burrowed soils and marine mammal haul outs – which also correspond to higher mouse densities around the fertile coastal zone and areas of interest for detecting mice, e.g. penguin colonies (Russell 2012; Elliott et al. 2015).

Four of the islands were not baited during the mouse eradication because we were confident mice were not present: Bollons Island (52.6 ha) and Archway Island (6 ha), together approximately 1.5 km to the north; West Windward Island (7 ha); and East Windward Island (8.5 ha) (Fig. 1). Evidence for this decision was based on absence of sign from inked tracking cards baited with peanut butter and placed inside corflute footprint tracking tunnels (FTTs; Black Trakka, Gotcha Traps, Rodney, New Zealand). During the operation six FTTs were used on Bollons Island, and 10 FTTs on each of the Windward Islands for 12 nights (Horn et al. 2019). The team could not land on Archway Island to monitor for mice but its distance from Antipodes Island, proximity to Bollons Island and the absence of sign of mice during previous monitoring (Marris 2000; Russell 2012) gave us confidence the absence of mice on Bollons Island was also analogous for Archway Island. Two other islands, Leeward Island (12.7 ha) and Orde Lees Islet (1.8 ha), were baited because we could not rule out the presence of mice due to their proximity to Antipodes Island and their inaccessibility for monitoring (Fig. 1).

Eradication confirmation

Monitoring to determine if mice had been eradicated occurred between 12 January and 15 March 2018 (late summer/early autumn), 18 months (including two summers) after the baiting operation. Monitoring was delayed this long after bait spread to allow any surviving mice to breed to an easily detectable level. Monitoring was timed for late summer when mice are breeding and juveniles have left the den (Russell 2012). Two primary methods were employed for mouse detection: FTTs and searching with dogs trained to detect rodents; with both methods supplemented by human observation.

The location of tracking tunnel transects was targeted to ensure accessible coverage of all habitat types, particularly adjacent to areas inaccessible to people and dogs, places where mice had been abundant before the eradication and areas where food was most available. All sites were recorded with handheld global positioning system (GPS) devices

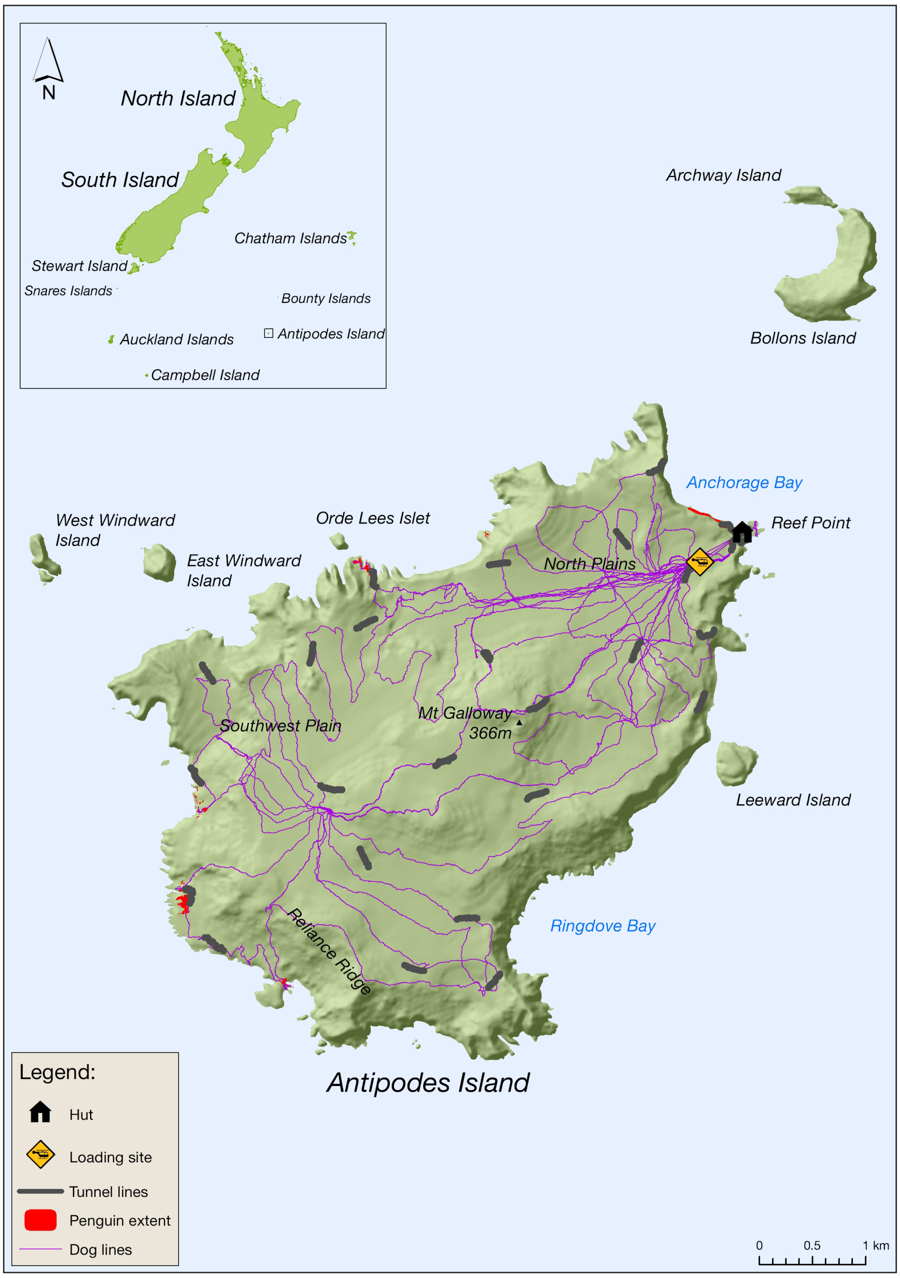

Figure 1. Map of Antipodes Islands group location; and mouse monitoring activity on Antipodes Island in 2018 showing the location of tracking tunnel transects, rodent searching with detection dogs, penguin colonies and field hut and helicopter hanger sites.

(Fig. 1). These sites included the area surrounding the field hut and penguin colonies (Fig. 1). Each transect was 200 m long with ten tracking tunnels spaced 20 m apart. A total of 270 tracking tunnels were installed. Inked tracking cards were baited with peanut butter and replaced approximately every 5 days over 3 weeks. Tracking cards were examined by experienced researchers to detect mouse sign and interference by non-target species. In response to anticipated interference all tracking tunnels had their entrance modified to exclude birds (reduced entrance size to c. 50 mm × 100 mm) and the tracking cards and tunnels were pinned to the ground (Fig. 2). Overall effort using tracking tunnels is summarised as 'tracking tunnel nights' (number of tracking tunnels deployed × number of nights deployed).

Two experienced rodent dog handlers and three rodent detection dogs (two border terrier cross fox terrier breeds and one Jack Russell) searched extensively for mice across Antipodes Island between 21 February and 15 March 2018. The dogs were trained and certified for rodent detection through DOC's Conservation Dogs programme. Two dogs had more than 3 years' experience post-certification. The third detection dog was newly certified and taken primarily for contingency. Guided by the field teams deploying tracking tunnels, handlers traversed all habitat types to accessible routes and areas of interest. Dog searches focussed on areas of short vegetation, exposed slips, and coastal areas particularly penguin colonies rich in food for mice (Elliott et al. 2015) (Fig. 1). Dogs were fitted with Garmin VHF collars and handlers carried Garmin (Lenexa, USA) Rhino handheld GPS units which displayed each other's position for coordinated searching and recorded daily track logs and areas of interest. Trail cameras were on hand to use where sign was ambiguous and further evidence was needed. Trail camera imagery was processed on site within the day of data cards being retrieved. All monitoring records were collated and reviewed by the monitoring team and DOC's Island Eradication Advisory Group (IEAG). The IEAG considered

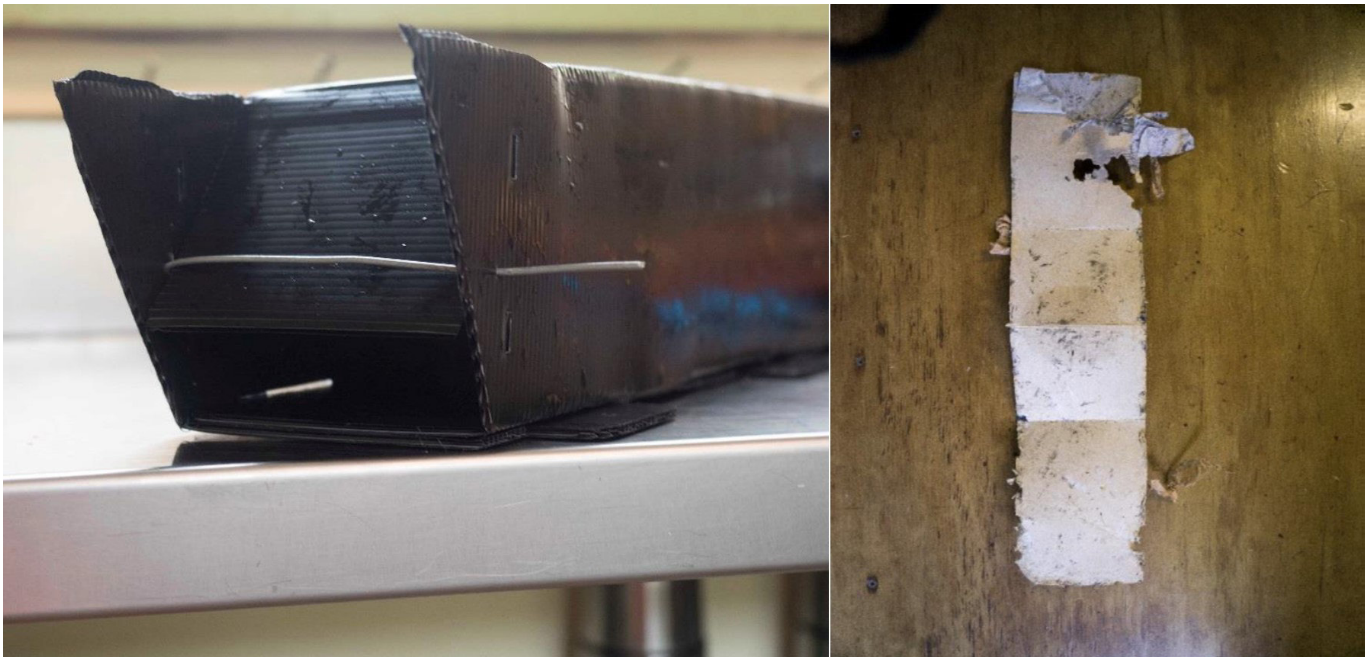

Figure 2. (A) Tracking tunnel with modified entrance (50 mm x 100 mm opening) to exclude birds; and (B) tracking card damaged by a parakeet (chewed edges).

the data, the delivery standard of the baiting operation and the time elapsed since baiting to advise on confidence in the eradication result to inform a formal declaration.

Monitoring data were analysed using rapid eradication assessment (REA) following Kim et al. (2020) via the REA interface (www.rea.is). Data were collated, including static device locations (footprint tracking tunnels; n = 270) and rodent dog tracks. The model was run using species and island specific parameters (Table 1). The reinvasion probability was set lower than other islands (e.g. Maud Island; Oyston et al. 2022) because the island is not publicly accessible, is distant from other islands, is visited rarely and is not used as an anchorage by fishing vessels. For comparative purposes the model was first run using only static devices (270 FTTs), then with static devices and rodent detection dog tracks.

Land bird monitoring

Land bird surveys on Antipodes Island were conducted annually in summer (December to February) by researchers visiting as part of a long-term population study of Antipodean wandering albatross. Additionally, surveys were undertaken opportunistically when researchers were present on the island to prepare for the eradication in August 2014 and October/November 2014; by the operational team pre-eradication in June 2016 and post eradication in August 2016; and between 21 February and 15 March 2018 by members of the mouse eradication result monitoring team. Baseline monitoring included the surveys between 2013 and June 2016; and impact and response monitoring included surveys conducted between

Table 1. Rapid eradication assessment model parameters used for Antipodes Island. Parameters derived from Russell (2012); Russell et al. (2017); Kim et al. (2020); Sagar et al. (2022).

Parameter	Likely	Min–max
Monitoring data		
Monitoring nights	30	
Iterations	2000	
Target	0.95	
Years since eradication	1.5	
Device parameters		
g_0 (tracking tunnels)	0.2	0.15–0.25
Biological parameters		
σ	13	10–16
Prior probability of success	0.8	0.7–0.9
Probability of reinvasion	0.005	0–0.01
Population growth rate (annual per capita)	7	5–10
Dispersal distance	50	
Incursion distance	200	

August 2016 and January/February 2022.

Snipe

Snipe abundance has been monitored between December and March every summer since 2012/13 for four summers before the mouse eradication and five summers after. Monitoring was undertaken by researchers working in pairs. Snipe counting was incidental to other work; workers recorded the amount of time spent away from the hut and the number of snipe seen. The metric of snipe abundance is the number of snipe seen per hour, with surveys lasting between 3 and 12 hours. Snipe that were heard but not seen were not recorded as there is considerable variation in snipe calling rates, with snipe almost silent in December and much of January, but much more vocal thereafter (Miskelly et al. 2006). Snipe counts were analysed using a generalised linear model with negative binomial errors and an offset for the length of survey time in the R package MASS (Venables & Ripley 2002). Mean snipe per hour was estimated for each summer along with testing for a change since the last summer.

Parakeets and pipits

Distance sampling (Buckland et al. 2001) was used to estimate the density and abundance of Antipodes parakeet, Reischek's parakeet and pipit. The perpendicular distance to individuals or groups of birds was measured from transect lines of variable length to the nearest metre ±0.5 m using a Nikon (Nikon Corporation, Tokyo, Japan) Forestry 550 Laser Rangefinder. Transects were distributed throughout the island and repeated as often as practicable. The aim was a sample of 60 to 80 encounters of each species for robust modelling of the detection probability and resultant population density. The technique relies on sightings of birds, so sampling was not undertaken in wet or cold weather when birds were likely to be less conspicuous. The computer software Distance 6.2 (Thomas et al. 2010) was used to analyse the data and compute population estimates. As the number of detections recorded was low for many of the survey periods, data were pooled and a global detection function computed from which survey specific estimates of density were calculated (Buckland et al. 2001; Thomas et al. 2010). Visual comparison of point estimates and their 95% confidence intervals were reinforced using a comparison of Poisson rates (poisson.test; R Core Team 2020) for three paired pre- and post-toxin application surveys and departures from a null hypothesis of no change in density were tested for.

Weed surveys and disturbed ground

Disturbed sites where ground was cleared for infrastructure and areas where materials were offloaded and stored were recorded using handheld GPS units. Between 2017 and 2022 researchers surveyed these previously operational sites annually in summer for weeds. Researchers knowledgeable in plant identifications used GPS records to guide survey boundaries. Weed plants found were recorded and removed where possible and referenced against historic records (Huggins 2016). Photos were used to monitor vegetation recovery at operational sites where major ground disturbances occurred. Photos were taken using a digital camera before and after at a site behind Reef Point that was cleared of vegetation for installation of the temporary helicopter hangar in June 2016. Follow-up photos were taken in August 2016 after the hangar was removed and to demonstrate recovery in March 2018 and again in February 2021.

Results

Mice

Tracking tunnels were deployed for a total of 7170 tracking tunnel nights and dog handlers with dogs covered 217 km of terrain with no evidence of mouse presence detected (Fig. 1). One rodent detection dog showed interest in an area of tight vegetation on Mt Galloway on 23 February 2018. The area was thoroughly investigated in response: an additional tracking tunnel transect was installed and run for 160 tracking tunnel nights and further dog searches undertaken with no sign of mice found. It was concluded that mice were not present in the area. The eradication of mice from Antipodes Island was declared by the New Zealand Minister of Conservation on 21 March 2018 (DOC 2018).

Rapid eradication assessment estimated a moderate to strong level of confidence in the eradication result when rodent detection dogs were used with static detection devices (50.5% above the 95% threshold), but limited confidence when static devices alone were tested (4.7% above the 95% threshold; Table 2). The modelling used suggests island coverage was greatly enhanced by using the rodent detection dog searching with static detection devices (24%), compared to detection devices alone (1.5%; Table 2). The probability of correctly declaring eradication success increased significantly when rodent detection dogs were used in conjunction with static devices (99.9%), compared to static devices alone (80.3%; Table 2). Non-target species regularly interfered with detection devices, removing bait from 75 tracking tunnels and inked tracking cards from 10. Pipits and parakeets were recorded on 83 and 43 inked tracking cards respectively and caused significant damage to many cards. Subantarctic skua (*Catharacta antarctica*) were also observed interacting with tunnels and removing cards.

Table 2. Rapid eradication assessment (Kim et al. 2020) model results for the island coverage of detection tools, median probability of eradication success and credible interval value (percentage of the posterior probability of eradication above the success target value; 95%) for Antipodes Island mouse eradication.

	Static devices	Static devices + rodent detection dogs
Coverage	1.5%	24%
Posterior probability of eradication success (2.5% and 97.5% quantiles)	80.3% (73.3–100%)	99.9% (73.9–100%)
Credible interval value (%)	4.7%	50.5%

Snipe

Snipe abundance was roughly stable for the first 3 years of baseline monitoring (2013 to 2016) and similar to results recorded in 2002 and 2003 (Miskelly et al. 2006). Snipe abundance declined abruptly just before the mouse eradication (Table 3; Fig. 3). There was no change in snipe abundance between the summers immediately before and after the mouse eradication, but in the following two summers snipe abundance increased dramatically. Snipe abundance has remained roughly stable from 2019 to 2022. Snipe are now 3 times more abundant than they were in the first 3 years of monitoring, and 10 times more abundant than they were in the two summers immediately before and after the mouse eradication (Table 3; Fig. 3).

Parakeets

Parakeet conspicuousness and abundance vary considerably with time of year so only counts carried out at similar times of year are comparable. This restriction means that the abundance estimates taken in winter 2016 before (June) and after (August) mouse eradication are comparable only with each other while those undertaken in January or February are comparable between years. Sampling in summer 2020 occurred in March after juveniles had fledged, inflating the result compared to January or February in other years, particularly for Antipodes parakeet and pipit. The population density of parakeet species decreased post-baiting compared to pre-baiting levels indicating impact from the baiting operation.

Table 3. Snipe surveys between the summers of 2012/13 and 2020/21, and the change in sighting rate between years (snipe per hour divided by snipe per hour in the previous year). P-value **<0.01, ***<0.001.

	Hours surveyed	Snipe seen	Snipe per hour	Change between years	P-value
2013	341.00	38	0.111		
2014	206.75	26	0.126	1.128	0.508
2015	140.50	17	0.121	0.962	0.926
2016	178.00	6	0.034	0.279	0.008 **
2017	224.00	8	0.036	1.060	0.935
2018	570.75	97	0.170	4.759	0.000 ***
2019	284.00	87	0.306	1.802	0.000 ***
2020	256.00	80	0.313	1.020	0.960
2021	314.50	101	0.321	1.028	0.818
2022	289.50	106	0.366	1.140	0.716

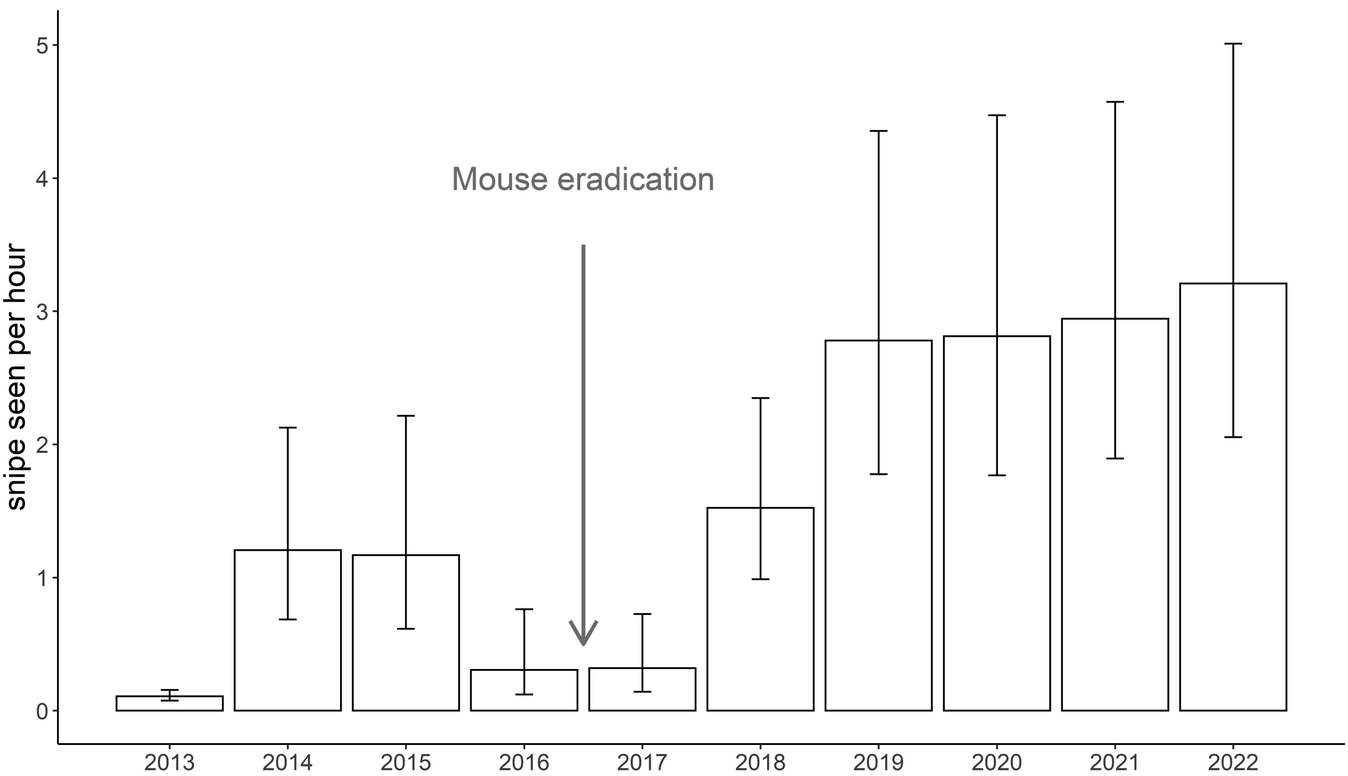

Figure 3. Snipe seen per hour in surveys between the summers of 2012/13 and 2021/22. Error bars represent 95% confidence intervals.

The population decrease was more severe for Reischek's parakeet than Antipodes parakeet. By January–February 2017 (the summer after toxin application), Reischek's parakeet had recovered to levels comparable to pre-baiting (2014) (Table 4; Fig. 4). Reischek's parakeet population density and abundance have increased steadily since 2016 and are now higher than the population size estimated prior to mouse eradication (Table 4; Fig. 4). The January–February 2017 survey results show the Antipodes parakeet population took longer to recover, reaching pre-baiting levels by 2018, two summers after toxin application. The population density and abundance are now similar to that estimated in 2019 and has possibly stabilised at slightly higher numbers than those seen prior to mouse eradication in 2014 (Table 5; Fig. 5).

Pipits

The population density of pipits reduced immediately post baiting but subsequently increased and appears to have begun to stabilise between 4 to 5 birds per ha. This density is much higher than the population density recorded prior to 2016 with very large year-on-year increases since 2016 (Table 6; Fig. 6).

Anecdotal observations

Revegetation of the cleared helicopter hangar site was rapid between 2016 and 2021 with native tussock and grass species covering the slopes within two seasons (Fig. 7A–7H). The introduced species *Hebe salicifolia* turned up at the hangar site and the introduced dock *Rumex obtusifolius* was found at the hut in the summer of 2017 after the eradication trip. Two exotic grasses were found in 2019, a single plant of sweet vernal grass (*Anthoxanthumodoratum odoratum*) turned up at the Castaway Depot steps and exotic creeping bent (*Agrostis stolonifera*) at one of the tent sites used in the result monitoring trip. All detected weed plants were removed and have not been detected again.

The endemic fly (*Xenocalliphora antipodea*) was obviously more abundant in January and February 2017 than it had been before eradication, and it was even more abundant in 2018 and 2019. It was less abundant in 2020 and 2021 although still more abundant than before the eradication.

In 2018 a large emergence of native noctuid moths was also observed (almost certainly *Graphania ustistriga*) by the field hut, and during that summer their big caterpillars, which

Table 4. Density estimates (95% CI) for Reischek's parakeets on Antipodes Island between 2013 and 2021.

Survey Date	n	Density ha^{-1} (CI)	Abundance (CI)
July 2013	29	2.3 (1.6–3.5)	4779 (3211–7113)
Feb 2014	46	2.1 (1.5–2.9)	4287 (3119–5894)
Aug 2014	9	3.7 (1.7–8.0)	7521 (3491–16 202)
Oct–Nov 2014	61	3.2 (2.4–4.3)	6478 (4828–8692)
Pre-drop Jun 2016	63	3.2 (2.4–4.4)	6569 (4825–8944)
Post-drop Aug 2016	173	0.6 (0.5–0.7)	1127 (921–1381)
Jan–Feb 2017	63	1.9 (1.5–2.6)	3930 (2946–5241)
Jan–Feb 2018	125	2.8 (2.3–3.4)	5608 (4590–6852)
Jan–Feb 2019	45	2.7 (2.0–3.8)	5496 (3976–7598)
Mar 2020	17	4.0 (2.4–6.6)	8035 (4854–13 302)
Jan–Feb 2021	89	4.8 (3.8–6.0)	9676 (7683–12 185)

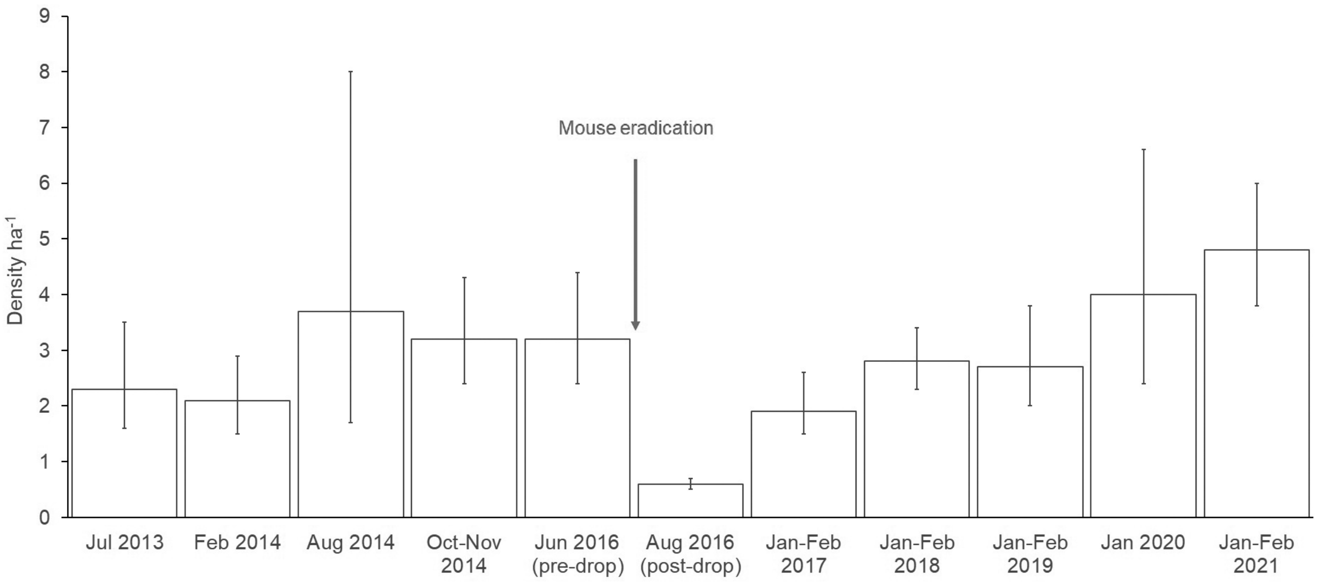

Figure 4. Density estimates for Reischek's parakeets on Antipodes Island between 2013 and 2021. Error bars represent 95% confidence intervals.

Table 5. Density estimates (95% CI) for Antipodes parakeets on Antipodes Island between 2013 and 2021.

Survey date	n	Density ha^{-1} (CI)	Abundance (CI)
July 2013	16	0.9 (0.5–1.7)	1817 (981–3366)
Feb 2014	37	1.5 (0.9–2.5)	3002 (1816–4964)
Aug 2014	7	2.3 (0.8–6.6)	4749 (1692–13 333)
Oct–Nov 2014	22	0.6 (0.4–1.1)	1275 (750–2168)
Pre-drop Jun 2016	22	0.5 (0.3–0.8)	922 (536–1584)
Post-drop Aug 2016	116	0.3 (0.2–0.4)	527 (360–772)
Jan–Feb 2017	31	0.7 (0.4–1.1)	1397 (852–2292)
Jan–Feb 2018	88	1.3 (1.0–1.8)	2683 (2004–3593)
Jan–Feb 2019	36	1.9 (1.3–2.7)	3761 (2548–5550)
Mar 2020	27	4.9 (3.2–7.9)	9895 (6460–15 157)
Jan–Feb 2021	50	1.7 (1.2–2.3)	3429 (2487–4727)

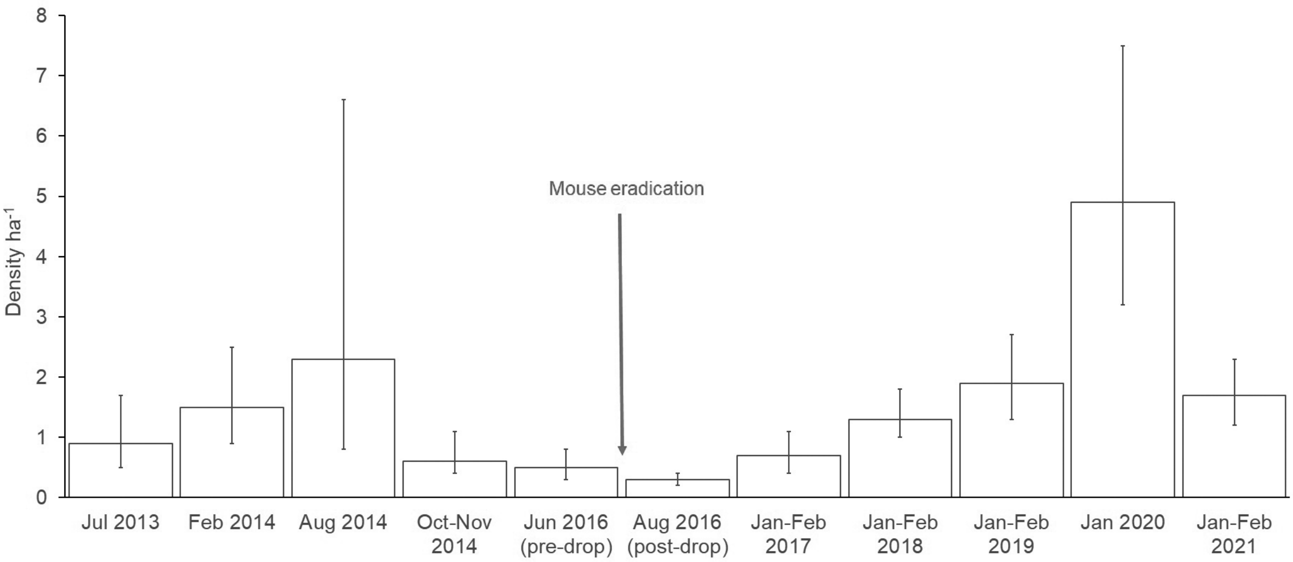

Figure 5. Density estimates for Antipodes parakeets on Antipodes Island between 2013 and 2021. Error bars represent 95% confidence intervals.

Table 6. Density estimates (95% CI) for pipits on Antipodes Island between 2013 and 2021.

Survey date	n	Density ha^{-1} (CI)	Abundance (CI)
July 2013	4	0.2 (0.1–0.5)	427 (164–1110)
Feb 2014	39	1.2 (0.8–1.8)	2480 (1658–3711)
Aug 2014	4	0.9 (0.4–2.4)	1867 (718–4857)
Oct–Nov 2014	108	1.2 (0.8–1.8)	2394 (1596–3591)
Pre-drop Jun 2016	101	3.2 (2.3–4.5)	6471 (4643–9020)
Post-drop Aug 2016	40	0.2 (0.2–0.3)	458 (332–632)
Jan–Feb 2017	62	1.1 (0.8–1.6)	2245 (1572–3205)
Jan–Feb 2018	227	4.4 (3.5–5.5	8904 (7146–11 094)
Jan–Feb 2019	82	5.2 (3.9–6.8)	10 441 (7882–13 830)
Mar 2020	25	6.9 (4.5–10.6)	13 905 (9030–21 414)
Jan–Feb 2021	87	4.4 (3.5–5.5)	8885 (7059–11 182)

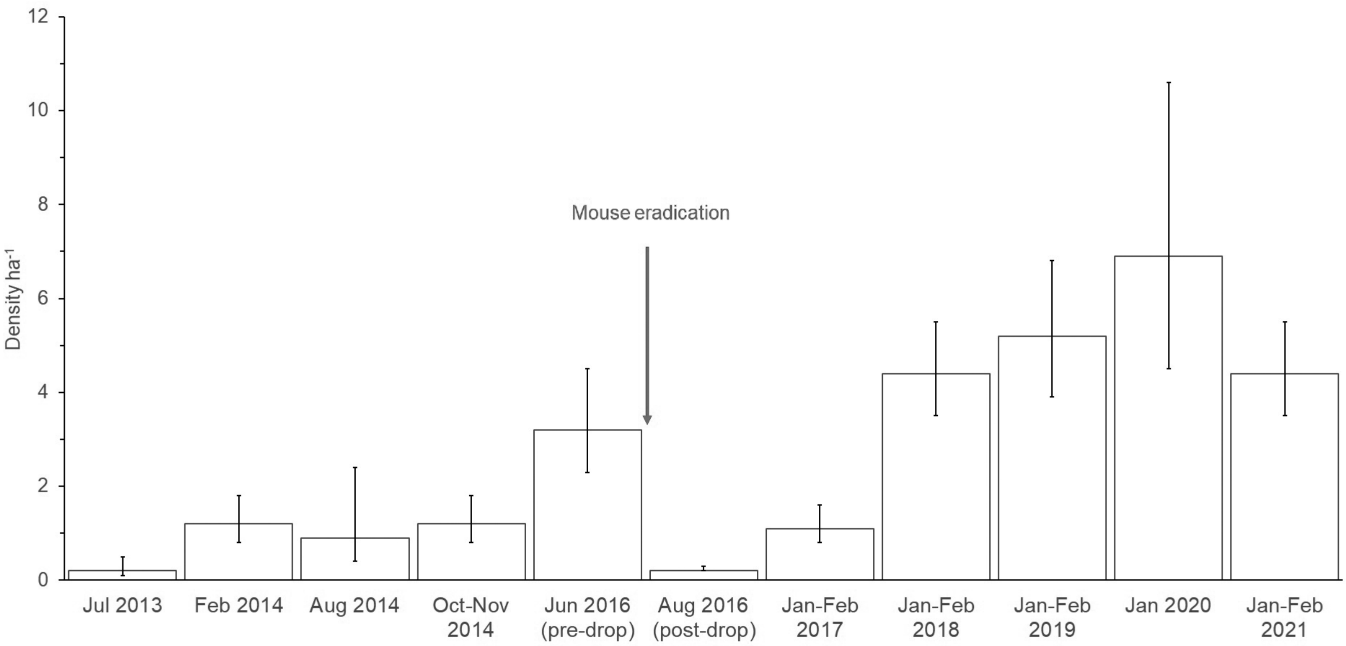

Figure 6. Density estimates for pipits on Antipodes Island between 2013 and 2021. Error bars represent 95% confidence intervals.

had only been observed once before, were conspicuous. The abundance of this moth has subsequently declined, although in 2021 it was still more abundant than it was before the eradication.

Native silvereyes (*Zosterops lateralis*) seem to have increased in abundance for a couple of years following the eradication. Pre-eradication, silvereyes were only encountered in tall forest of *Coprosma rugosa* just below the Dougal Stream waterfall, and in *Coprosma* spp. near Pipit Peak. In 2018 they were widespread and abundant, although still uncommon. By 2021 their numbers appear to have returned to levels a little higher than pre-eradication and they are more widespread.

Introduced mallard ducks (*Anas platyrhynchos*) were not readily observed following baiting but by 2018 they were back to previously observed levels (GPE and KJW pers. obs. 2021).

Introduced dunnocks (*Prunella modularis*) were very uncommon before eradication and would only be seen or heard as infrequently as once a week during periods of observation in summer. By 2021, many tens of dunnocks were seen and heard every day, and they were more common than pipits.

Discussion

Confidence to declare the eradication of mice successful in 2018 was based on a range of factors: successful completion of baiting in 2016; the absence of mice detections during annual summer visits to the island between 2016 and 2018; the nil result from rodent monitoring with people, dogs and devices in 2018; the REA model prediction that there was a high probability that there were no mice; and the recovery of mouse vulnerable wildlife (Horn et al. 2019). Had eradication failed, it is highly likely mice would have been detected during surveillance >2 years following the eradication attempt (Broome et al. 2019). Confidence has been further increased by the absence of mouse sightings or detection of their sign in static detection devices and around the hut during annual 2-month long visits to the island between 2018 and 2022.

The REA models suggest that rodent detection dogs increase the island coverage and therefore confidence in declaring eradication success, especially for larger islands (e.g. Great Mercury Island; Kim et al. 2020). While helpful for estimating confidence in eradication success, this model is not the sole means of determining eradication success. For example, it does not incorporate the absence of detection of mice by visitors to the island not involved in checking tracking tunnels or working with rodent detection dogs. However, use of this model in different environments, with differing device characteristics and focal species parameters, increases the functionality and realism of the model for future use. REA modelling indicates that assessing eradication success soon after an eradication operation (Russell et al. 2017) would have not been achievable at Antipodes Island because of the high intensity of monitoring required (grid spacing of devices <60 m), the scale of the island and large areas of inaccessible terrain. It would also be of limited value as the difficulty of getting to Antipodes Island means that it would not have been possible to quickly mount an operation to target any survivors detected.

Chew-cards and wax-tags are regularly used for rodent abundance index surveys (Sweetapple et al. 2006; Wilmshurst & Carpenter 2020), but tracking tunnels were the only detection device used on Antipodes Island to minimise identification ambiguity. There is a high level of confidence identifying mice footprints, but bite marks can be hard to distinguish (Thomas 1999; Olivera et al. 2010). On Macquarie Island, slug gnaw marks on wax tags were very similar to that of mice (SRH, pers. obs. 2012) and on Adele Island, New Zealand wētā created marks on wax tags that confounded results of the survey for mice (Livingstone et al. 2022). Given individual rodents vary in their susceptibility to detection devices (Wilmshurst & Carpenter 2020) the use of detection trained dogs and human observations were an important contribution to the confidence that non-detection of mouse sign equated to complete absence of mice. The detection dogs were all small bodied short-legged breeds and they struggled in areas of tall *Poa litorosa* vegetation and *Polystichum* fern, which also hindered humans.

Figure 7. Vegetation disturbance and recovery at the temporary helicopter hangar site on Antipodes Island. Images shown in order: (A) field hut and historic castaway depot; (B) field hut and historic castaway depot post landslip January 2014; (C) hangar site pre-clearance 28 May 2016; (D) hangar site post-clearance and during construction; (E) hangar site with hangar completed 5 June 2016; (F) hangar site following removal of the hangar 3 August 2016; (G) hangar site with tussock cover 3 March 2018; (H) hangar site 5 February 2021 (Photos: A, B & H – KJW; C, D, E & F – SRH; and G – FSC).

Dog handlers would lose visual contact with dogs in these areas, which influenced the habitat searched. These constraints highlight the value of the delay between baiting and monitoring to increase the probability of detecting a failed eradication. This delay is particularly important where large areas of habitat or terrain are inaccessible. Future result monitoring on islands with similar habitat and vegetation classes should consider inclusion of some longer-legged detection dogs.

Pipits and parakeets likely contributed to the 10 cards removed from tracking tunnels based on the evidence on the cards (FSC, pers. obs. 2018). Although non-target interference with tracking tunnels was manageable with the modifications employed (Fig. 2), it is likely that non-target interactions with accessible chew card and wax-tags would have been greater had they been used. The tracking tunnels were physically bulky to deploy and with the need to reduce entrance size, prefabricated smaller tunnels would have been better. For large-scale mouse eradications where other rodent species are not present, such as Auckland Island, smaller tunnels should be considered. As part of feasibility trials on Auckland Island, a prototype (50 mm × 50 mm × 500 mm) was tested and showed no significant difference in detection probability for mice when compared to conventional tracking tunnels but were a quarter of the size of standard tracking tunnels to carry and deploy (Cox & Ware 2020).

Prior to the mouse eradication, snipe were much less abundant on Antipodes Island than they were on pest-free Adams Island in the Auckland Islands (Miskelly et al. 2006). Following the elimination of mice, snipe numbers have significantly increased on Antipodes Island. It seems reasonable to conclude that through competition for invertebrate food mice were suppressing snipe abundance on Antipodes Island and the elimination of mice has allowed the snipe population to increase. Prior to the mouse eradication snipe populations were low, mostly stable but occasionally suffering declines such as the one before the mouse eradication between 2015 and 2016. The lack of population change between the summers before and after the eradication suggests that there was no dramatic decline caused by the baiting. Following mouse eradication, observations for snipe grew between 2017–2019, and have remained stable between 2019–2022 at more than double the observed pre-eradication rate. The snipe population is likely approaching a new equilibrium on Antipodes Island and it is interesting that this new density is still much lower than on Adams Island (Miskelly et al. 2006, 2020) where rodents have never established. Snipe are particularly abundant and/or conspicuous on Adams Island at high altitudes where rainfall is high and soils are moist (GPE and KJW pers. obs. 2021); a habitat almost absent on Antipodes Island, which is much smaller, more exposed and less diverse than Adams Island.

Following initial losses from baiting, pipit and parakeets recovered quickly. Pipit and Reischek's parakeet populations surpassed pre-eradication levels within two summers and have responded strongly to the removal of mice. The recovery and stabilisation of the Antipodes parakeet population also featured year-on-year increases for the first three summers following baiting. Pre-eradication bait uptake trials on Antipodes Island indicated risk to pipits from primary poisoning but not for parakeets (Elliott et al. 2015). During operational design, captive management was trialled for parakeets and considered for pipits, parakeets and snipe as a mitigation for the risk of by-kill (Elliott et al. 2015). Captive management was not pursued because populations of pipit and parakeets on nearby pest-free Bollons and Archway Islands, where bait would not be spread, were deemed large enough to sustain and restock Antipodes Island if impacts exceeding predictions occurred. A similar pattern of short-term impact, rapid recovery and significant population increase above pre-eradication levels was recorded for Berthelot's pipit (*Anthus berthelotii*) on the Macranesian island of Selvagem Grande when mice and rabbits were eradicated (Olivera et al. 2010). Pipit (*Anthus antarcticus*) on South Georgia were rare because of predation by rats, but also quickly re-established and bred successfully in rodent-free areas following rat eradication (Martin & Richardson 2017). Ground nesting land birds on Hawadax Island had strongly recovered 5 years post-eradication (Croll et al. 2016).

Ducks are susceptible to bait consumption and poisoning during rodent eradications (Dowding et al. 1999, 2006; Eason et al. 2002). One mallard duck was found in 2016 that had succumbed to bait and mallards may have been eradicated and recolonised, but more likely they were almost eradicated and recovered. In January/February 2021, tens of dunnocks were being seen per day (GPE and KJW, pers. obs. 2021) compared with up to five per day witnessed between late April and early June 2001 (Imber et al. 2005). We postulate that once the mice were gone small passerines like the dunnock, silvereye and pipit started to increase in abundance in response to the abundant invertebrate food. However, after a couple of years the dunnocks were so abundant that they once again suppressed the flies and moths, and the silvereyes were out-competed back to close to their original abundance.

A focussed approach was required for ecological monitoring on Antipodes Island. The remoteness and isolation combined with steep topography, dense vegetation and variable weather meant that access was limited by difficult logistics and high cost, constraining the extent and timing of investigations. Annual visits to undertake a long-term study of Antipodean albatross enabled post-operational monitoring of indicator land birds and seabirds at low cost. Occasional inconsistency in seasonal timing was unavoidable some years because of logistical constraints and caused some results to be less comparable. Baseline studies for land birds were also unable to include control sites. Pest-free offshore islands in the Antipodes Islands group large enough to provide scientific controls (Bollons and Archway Islands) were not logistically feasible to access because they can only be visited when rare periods of calm seas and calm weather coincide with the start or end of an expedition when boat transport is present. Despite these monitoring constraints, strong positive trends were measured for indicator species that were vulnerable either to baiting or suppression by mice. This monitoring effort adds to the growing body of evidence of the impact of mice on islands (Dilley et al. 2015, 2016, 2018; Broome et al. 2019) and that they should not be overlooked when considering multi-species eradications from islands.

On Antipodes Island good baseline measurements exist for some seabirds (Imber et al. 2005) and invertebrates (Marris 2000; McIntosh 2001; Russell et al. 2020b) and these provide an opportunity for further investigations of longer-term outcomes of eradicating mice. Seabird species for future monitoring could include the important population of grey petrel, subantarctic little shearwater, grey-backed storm petrel and black-bellied storm petrel. Storm petrels were likely supressed by mice on Antipodes Island (Imber et al. 2005; Martin & Richardson 2017) although a small number of observations of Antipodes parakeet actively hunting grey-backed storm petrel (Greene 1999) suggest mice were not the only predatory influence on their breeding. Long-term monitoring of these potentially slow recovering seabirds will be informative for eradication of mouse, cats and pigs from Auckland Island where the seabird fauna is depauperate due to suppression by these mammals (Miskelly et al. 2020; Russell et al. 2020a).

Biosecurity management was rigorous for the extraordinary amounts of cargo and personnel that went to Antipodes Island for the eradication operations (Horn et al. 2019) but some weed seeds still arrived and grew. Annual weed observations and management were a fortunate benefit of the annual presence of albatross researchers as funding for the eradication was limited to the sowing of bait and result monitoring in 2018. The containment of unintentional weed spread highlights the importance of investing in targeted surveillance for an extended period after intensive operational activity. Commitment to surveillance and monitoring for understanding outcomes and unintended consequences requires clear definition of objectives. Funding for monitoring for future projects could be separated from eradication funding to reduce the likelihood of reprioritising resources after an eradication operation is complete (Bird et al. 2019).

Monitoring the success of a mouse eradication two summers after the baiting operation achieved high confidence with moderate effort using a simple monitoring regime comprising human observation, detection dogs and a single detection device (tracking tunnels). Eradicating mice has improved the conservation value of Antipodes Island and ecosystem recovery has been occurring without further intervention. Pre-eradication monitoring informed decision making for operational design and helped manage operational risks. Post-eradication monitoring confirmed anticipated outcomes. Targeted monitoring of native land birds, vulnerable to predation by rodents, was an efficient indicator of operational impacts and ecosystem recovery following rodent eradication in this remote and isolated place. The strong response of snipe, pipit and Reischek's parakeet highlights their previous suppression by mouse presence. There were no negative long-term impacts from the operation with rapid recovery of disturbed vegetation and land bird species that suffered non-target mortality. Captive management was not required because pest-free offshore islands provided refuge for viable populations of native land bird species to mitigate unanticipated operational risk. Anecdotal observations suggest there was an initial flush of invertebrates following mouse eradication before the predatory void left by mice was taken over by other bird species including exotic passerines. Eradication has proven to be an effective one-off action for reversing the chronic damage done by mice on Antipodes Island.

Author contributions

SRH, FSC, GPE, KJW, JCR and TCG designed the study and undertook fieldwork; GPE, TCG, JCR and RLS analysed the data; and SRH, GPE and TCG wrote the manuscript with input from FSC, JCR and RLS.

Acknowledgements

Thank you to the monitoring teams that contributed to this work and the support of the Murihiku district team. Thank you to Steve Kafka and the crew of the MY Evohe, for transporting teams to and from Antipodes Island. Thank you to Nick Holmes and Elizabeth Bell for helpful reviews of this paper.

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Editorial board member: Thomas Etherington
Received 15 March 2021; accepted 17 October 2022