The effects of introduced mice on seabirds breeding at sub-Antarctic Islands

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Declaration

This thesis reports original research that I conducted under the auspices of the FitzPatrick Institute, University of Cape Town. All assistance received has been fully acknowledged. This work has not been submitted in any form for a degree at another university.

Signed by candidate

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Ben J. Dilley

Cape Town, June 2018

A 10 day-old great shearwater *Ardenna gravis* chick being attacked by an invasive House mouse *Mus musculus* in an underground burrow on Gough Island in 2014 (photo Ben Dilley).

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Abstract

Seabirds play keystone roles as apex predators in marine ecosystems and also influence the ecology of terrestrial ecosystems where they breed. Seabirds are among the most threatened group of birds - almost half of all seabird species are known or suspected to be experiencing population declines with 97 (28%) of the 346 species currently classed as globally threatened and at risk of extinction. Introduced predators at oceanic islands where many seabirds breed account for the largest proportion of population declines, more so than incidental fisheries bycatch or degradation of their breeding habitats. Since few oceanic islands have escaped invasion, the problem is widespread, with the prime culprits being introduced cats *Felis catus*, rats *Rattus* spp. and house mice *Mus musculus* which depredate adult birds, chicks and eggs. Rats were widely introduced to thousands of islands and their catastrophic effects on seabird populations have been well documented. Mice are estimated to have invaded more oceanic islands than any other alien predator, but until fairly recently they were considered to have little impact on seabird populations.

This thesis focuses on seabirds breeding at two large oceanic islands - Marion Island (293 km^2) in the south Indian Ocean and Gough Island (65 km^2) in the south central Atlantic Ocean. Both islands have mice as the sole introduced mammal. Of relevance to this study, however, is that the density of burrow-nesting petrels is much higher on Gough Island because Marion Island's petrel populations were greatly reduced by cats, which were introduced in 1948 and eradicated by 1991. In the early 2000s, researchers on Gough Island identified mouse predation as the most probable cause of the high chick mortality of at least three species of seabirds, including the endemic Tristan albatross *Diomedea dabbenena*. Further research concluded that mice can be devastating predators of seabirds on islands where they are the sole introduced mammal, because in the absence of competition and predation from larger introduced species, mice can attain very high population densities, and resort to attacking seabird chicks mainly in winter when there are few other food sources. In 2003, the first mouse-injured wandering albatross *Diomedea exulans* chicks were found on Marion Island and in 2009 the first attacks on summer-breeding albatross chicks were recorded, but incidents appeared to be infrequent. Although mouse predation had been identified as a potentially serious threat to seabirds at both islands, further evidence was required on how many seabird species were being affected and to quantify the impacts. Field observations suggested a noticeable increase in levels of mouse predation at both islands, yet there was still no direct evidence of mice depredating burrow-nesting petrels at Marion. In this thesis I assess the impacts of invasive mice at both islands and establish pre-eradication baseline estimates for the burrow-nesting petrel populations at Marion Island.

Burrow-nesting petrels are the most abundant seabirds in the Southern Ocean, yet their populations are poorly known compared to surface-breeding albatrosses because they are difficult to survey accurately. Extrapolation from density estimates can lead to large error margins, but these can be reduced with the development of repeatable, island-specific survey methods for long-term monitoring. This forms the basis of **Chapter 2**, where I test the effect of sampling strategy (random transect or systematic survey) on population size estimates of three burrow-nesting petrel populations at Marion Island. Systematic, island-wide surveys were appropriate to estimate the population sizes of blue petrels *Halobaena caerulea* (strongly clustered distribution - **Appendix 1**) and white-chinned petrels *Procellaria aequinoctialis* (moderately clustered distribution - **Appendix 2**) and but for the very widely distributed great-winged petrels *Pterodroma macroptera* I counted burrows within random transects and extrapolated burrow densities by associated habitat attributes to generate island-wide estimates. The systematic surveys required more effort, but resulted in more accurate estimates for species with clustered distributions, whereas the random transects required less effort but resulted in broad estimates with wide error margins which limits the ability to detect changes over time.

In **Chapter 3,** I investigate how burrow-nesting petrel populations on Marion Island have recovered since cats were eradicated in 1991. In theory, the removal of cats as the superpredator, combined with endogenous growth and the potential for immigration from nearby mouse-free Prince Edward Island, could have promoted a multi-fold increase in petrel numbers over the last two decades. To investigate this, I repeated a burrow-nesting petrel survey in the north-eastern sector of Marion Island originally conducted by Mike Schramm in 1979 and assessed how burrow densities have changed compared to densities at the peak of the cat-era. I found that burrow densities have increased by a modest 56% since 1979. The recovery of summer-breeding petrels decreased with decreasing body size, and winter-breeding species showed even smaller recoveries, which is similar to patterns of breeding success at Gough Island where mice are the major drivers of population declines among petrels. Mice are the likely cause of the limited recovery of burrowing petrels at Marion Island.

To assess and document the impacts of invasive mice at both islands, I installed infra-red video cameras into burrows and assessed breeding success with regular burrow-scope nest inspections of study colony nests at both Gough and Marion Islands (**Chapters 4** and **5**). The results show that mice can be very effective predators of burrow-nesting petrel chicks and to a lesser extent, eggs. The breeding success for winter breeders were lower than for summer breeders at both islands, and among winter breeders most chick fatalities were of small chicks less than 14 days old. Fatal mouse attacks on small chicks were video recorded for six burrow-nesting petrel species and winter breeders had very high chick mortality rates (e.g. 82–100% on Gough Island). Since mouse depredation of seabird chicks was first identified as a problem in 2001, the frequency and severity of mouse predations appears to have escalated on Gough (**Appendix 3**), yet on Marion Island detected incidents remained infrequent until 2015, when mice attacked 4.0–4.6% of the large chicks of all three albatross species that fledge in autumn. Attacks started independently in small pockets all around the island’s 70 km coastline, separated by distances hundreds of times greater than mouse home ranges. Attacks have continued from 2016–2018 at varying rates on summer-breeding albatross fledglings, showing how mice alone may significantly affect threatened seabird species (**Chapter 6**).

In summary, mice appear to be suppressing the productivity of burrow- and surface-nesting seabird populations at both islands and are very likely causing population declines, especially among winter breeding species. Fortunately, the removal of invasive mice from islands through aerial spreading of toxic bait is a viable option and the scientific and visual evidence collected during this thesis has contributed to the growing body of evidence needed to persuade funders and Governments to support eradication operations at both study islands.

Acknowledgements

When I first went to Marion Island in 2009 I never imagined it would turn into an epic nine-year adventure of multiple expeditions to remote islands! It has been, and continues to be, an incredible experience and there are so many people to thank. First and foremost I would like to thank Peter Ryan and Delia Davies.

Thank you Peter for being my supervisor, mentor and friend; we've shared some great times on and off the islands and it's been a real privilege to learn from you. Thanks for making it all happen, for your clarity of thought and for your brilliant editing of drafts. Even when expensive burrow cameras got shredded by mice in the first weeks in the field, or when our wind turbine fell to pieces in Marion's gales, you didn't lose faith and we found new solutions.

Thank you Delia for being my constant companion to explore these remote islands, hike in the wind and rain, and live in huts and tents for months at a time in quite literally the middle of nowhere. It's been quite an adventure and I thank you for your love, generosity, insight, courage, your willingness to live simply in the sticks and for keeping us on the right track all these years. I look forward to the next island marooning!

I am grateful to Mike Schramm for having the foresight to survey the burrowing petrels back in 1979, and for then returning to Marion in 2012 to show me how it's done, how to wield a 'Ladies' Lasher' spade, operate a Schrammcam and for the continued friendship and much needed reality checks. To Marthàn Bester, thanks for you encouragement and dry humour on all the voyages, it was never boring! I commend you and all the cat hunters for successfully eradicating cats from Marion - without your perseverance we would have few burrowing petrels to count today.

To my team members on the Marion 66th and Marion 69th over-wintering expeditions (14 months) I extend huge thanks to all of you. My special thanks Mia Wege for all your support and friendship on both teams since 2009, you even got to Nightingale! Thanks Fred Fourie for your friendship on M69 and now in Simons Town and for the countless 05h30 ubers. For the burrow cameras, I thank Rory Meyer for helping sort the crucial LED burrowcam lights and to Nigel Butcher at RSPB for the MemoCams.

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Overwintering on sub-Antarctic Islands is expensive and logistically very complex. I thank everyone at the South African National Antarctic Programme (SANAP, Department of Environmental Affairs) and the crews of the *SA Agulhas* and *SA Agulhas II* for their support and dedication. Without their support, the three overwintering expeditions which I enjoyed and gained so much from would never have happened. Financial support for this study and the broader seabird projects on Marion was provided by SANAP and the FitzPatrick Institute of African Ornithology Centre of Excellence (CoE) through the National Research Foundation. Long-term monitoring on Gough Island was established with a grant from the UK Foreign and Commonwealth Office with further support over the years from the UK Government's Overseas Territories Environment Programme (OTEP) and Darwin Initiative programme, the Royal Society for the Protection of Birds and the Agreement on the Conservation of Albatrosses and Petrels.

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Chapter 1: General introduction: Islands, mice and seabirds

> *"A few domestic cats have gone feral [on Marion] and prey on the smaller petrels or mice that are widespread over the coastal plain"*
>
> (Rand 1954, p 178)
>
> *"Mice often burrow into the [albatross] nest cone but do no appreciable damage"*
>
> (Rand 1954, p 189)
>
> R.W. Rand, Biologist, Marion Island, 1951

The effects of invasive species on global biodiversity have been described as "immense, insidious and usually irreversible" (IUCN 2000). In the late 18th and early 19th centuries, humans travelled far and wide in the southern oceans to exploit marine wildlife (Trathan and Reid 2009) and an unfortunate consequence of this travel was the deliberate or incidental introduction of alien animal and plant species to distant and isolated environments, far beyond their normal geographical range, causing extensive changes in biological communities (Mooney and Cleland 2001). The severity of the invasive species' impacts varies depending on the evolutionary history of the community being invaded. Since island biotas have evolved as insular communities they are particularly susceptible to change (Warren *et al.* 2015). Few islands escaped introductions and invasive species are the main cause of species extinctions on islands (Manne *et al*. 1999; Chapin *et al*. 2000). Birds are a good case in point – 88% of the 140 bird species that have gone extinct since 1500 were endemic to islands (Butchart *et al*. 2006).

Oceanic islands and seabirds

Most oceanic islands are summits of massive shield volcanoes that rise up from the abyssal depths of the ocean (Baker *et al*. 1964). Unlike most coastal islands, oceanic islands have never been linked to a continental landmass and as such the ecosystems, flora and fauna on oceanic islands are particularly vulnerable to invasion and have disharmonic ecosystems that lack many elements of continental biota (Blackburn *et al*. 2005). The few animals and plants that manage to reach oceanic islands and survive there have evolved as insular species (species which exist in isolated natural communities). These species are at a distinct disadvantage when new (continental) predators or competitors are introduced by humans, since they have not evolved, or in some cases not retained (e.g. ability to fly; McNab 1994) the behavioural responses needed to cope with these new predators/competitors (MacArthur and Wilson 2001; Whittaker and Fernández-Palacios 2007).

The life-history characteristics of seabirds are typical of highly 'K-selected' species, having a long life expectancy, low adult mortality, delayed reproductive maturity, low annual productivity, small clutch size and extended fledgling periods (Stearns 1992; Tickell 2000). These attributes make seabird populations particularly vulnerable to predation from introduced predators, especially the continued loss of breeding adult birds, which can result in a rapid decline in the breeding stock. Historically, oceanic islands provided ideal safe breeding grounds for many seabirds because the only native predators are other seabirds such as skuas and gulls which prey on burrow-nesting petrels, eggs and chicks (Ashmole 1963). By contrast, predation by non-native (introduced or alien) mammals is far more devastating to seabird populations because, as ground predators, alien mammals are able to tackle all seabird life-stages (for example rodents, mustelids *Mustela* spp., mongooses *Herpestes* spp. and domestic cats *Felis catus* raid petrel nests which are normally protected in burrows). The life history traits of these seabirds are not adapted to cope with such predation: pairs seldom re-lay if they lose an egg or chick and it can take several years for a widowed adult to form a new pair bond (Chastel *et al*. 1995; Barbraud *et al.* 1999; Mauck and Grubb 1995). Because islands have a disproportionate share of global terrestrial biodiversity (Croxall *et al.* 2012; Spatz *et al.* 2017), introduced mammals are thought to have been major drivers of biodiversity loss, causing extinctions of insular endemic birds and local extinctions of island breeding seabirds (e.g. Veitch 1985; Medway 2004). In this regard, cats have been universally disastrous for many seabird populations since their predation of adult birds can decimate seabird colonies in mere decades (Keitt *et al.* 2002; Medina *et al*. 2011).

Overview of seabird threats and invasive species on oceanic islands

Seabirds forage at sea and breed on land, and so are reliant on two habitats for their survival. Some species also migrate, and thus require sea habitat in breeding and non-breeding areas. Seabirds are among the most threatened group of birds, with almost half of all seabird species known or suspected to be experiencing population declines: 97 (28%) of the 346 species are globally threatened and at risk of extinction (including 17 of the 22 species of albatrosses; IUCN 2018). Seabirds play keystone roles as apex predators in their marine ecosystems where they forage, and influence the ecology of terrestrial ecosystems (e.g. as nutrient providers) where they breed, and thus their population declines are of global conservation concern.

Several populations of albatrosses, petrels and shearwaters (Procellariiformes) have been detrimentally affected by at-sea threats, notably fisheries interactions such as competition for food (Cury *et al*. 2011), accidental mortality on fishing gear (Nel *et al*. 2002; Barbraud *et al*. 2009; Petersen *et al*. 2009), climate change impacting regional productivity (which are especially problematic for central-place foragers such as breeding seabirds, Krüger *et al*. 2018) and prey distribution (Grémillet and Boulinier 2009; Barbraud *et al.* 2012), plastic pollution (Wilcox *et al.* 2015; Ryan 1987; Pierce *et al.* 2004; Ryan 2008; Eriksen *et al.* 2014), oil pollution (Burger and Gochfeld 2002; Votier *et al.* 2008) and the effects of meavy metals in the marine environment (Muirhead and Furness 1988; Walsh 2017).

However, land-based threats remain a significant problem for seabirds and invasive mammals are the major drivers of extinction and ecosystem change on many oceanic islands. Seabirds breeding on islands are also particularly susceptible to multiple human-induced threats such as over-exploitation, disease (Weimerskirch 2004; Wikelski *et al*. 2004), invasive plants displacing native species (Reaser *et al*. 2007) and introductions of invasive organisms (Manne *et al*. 1999). Some introduced mammals (e.g. rabbits *Oryctolagus cuniculus* and goats *Capra hircus*) indirectly affect the birds through habitat destruction (erosion and burrow trampling) and habitat alteration (overgrazing). However, the introduction of mammalian predators such as pigs *Sus scrofa*, cats, rats (ship or black rats *Rattus rattus*, Norway rats *R. norvegicus* and Pacific rats *R. exulans*) and house mice *Mus musculus* to oceanic islands where seabirds breed directly affects birds through predation of eggs, chicks and, in some cases, adults, and this is *the* major threat to seabird species (Croxall *et al*. 2012). Since few oceanic islands have escaped invasion, introduced predators account for the largest proportion of seabird population declines, more so than incidental bycatch and competition for prey with commercial fisheries (Jones *et al*. 2008).

Rats were widely and (mostly) unintentionally introduced to thousands of islands, and the catastrophic effects on seabird populations through predation of chicks and eggs has been studied extensively (Atkinson 1985; Jones *et al*. 2008, 2012), including a number of global reviews highlighting the extent of the problem and which species are most at risk (e.g. Croxall *et al*. 2012; Medina *et al.* 2011; Ruffino *et al.* 2009; Harper and Bunbury 2015; Caut *et al.* 2007, 2008; Harris 2009; Towns *et al.* 2009; Fukami *et al.* 2006; Mulder *et al.* 2009; Brooke *et al.* 2010). However, despite mice being introduced to more oceanic islands than any other alien mammal (including rats; Moors and Atkinson 1984), and although their impacts on sub-Antarctic island biota are numerous (Angel *et al*. 2009), until fairly recently mice were considered to have little impact on seabird populations (Jones *et al*. 2003; Cuthbert and Hilton 2004; Wanless *et al*. 2007). Across most of their non-native range, mice are seldom the only introduced species, and thus the impacts of mice on island biota are difficult to separate from those of other introduced mammals (c.f. Angel *et al*. 2009).

Recent research by Cuthbert and Hilton (2004) and Wanless *et al*. (2007) found that mice can be devastating predators of seabirds when they are the only introduced mammal and how, in the absence of competition and predation from larger introduced mammals, mice can attain very high population densities, and resort to attacking seabird chicks, especially in winter when other food resources are scarce.

There are at least eight large oceanic islands which have mice as the sole introduced mammal and also support breeding colonies of seabirds (see Fig. 1.1. and Table 1.1): (1) Marion Island (cats were eradicated by 1991, but mice remain; Bester *et al*. 2000); (2) Antipodes Island (although mice were eradicated in 2016, https://www.islandconservation.org); (3) Selvagem Grande Island; (4) Sand Island, Midway; (5) Gough Island (where mice are the sole alien mammal to have ever been introduced); (6) St Paul Island (ship rats and rabbits *Oryctolagus cuniculus* were eradicated by 1999, but mice remain (Micol and Jouventin 2002); (7) Australie and (8) Mayes Islands in the Kerguelen archipelago (Micol and Jouventin 2002).

Incidents of mice depredating seabirds have been reported at five (Table 1.1) of these oceanic islands. However, with the exception of Gough and Marion, all the incidents were suspected predation events. These were based on observations of egg shells with incisor marks, chicks with wounds or mice seen feeding on chick carcasses. The possibility that mice were simply scavenging off discarded eggs or dead chicks cannot be ruled out. However, compelling evidence of much higher petrel breeding densities at mouse-free offshore stacks than on the mouse-infested main islands at some locations (e.g. black-bellied storm petrels *Fregatta tropica* on Antipodes Island, Imber *et al*. 2005) further points towards mouse predation.

Table 1.1. Documented incidents of suspected or confirmed cases of introduced house mice *Mus musculus* depredating seabird eggs and chicks on large oceanic islands (see Fig. 1.1 for locations).

Oceanic Island	**Year(s)**	**Species depredated by mice**	**Prey mass range**	**Extent & impact**	**Other introduced mammals**	**Source**
Marion Island	1982	Blue petrel *Halobaena caerulea*	eggs & chicks (30–100 g)	extent unknown	cats	a
	2003-2009	Wandering albatross *Diomedea exulans*	chicks (500 g – 5 kg)	12 cases, widespread	[cats <1991]	b
	2009	Sooty albatross *Phoebetria fusca*	chicks (500 g – 2 kg)	8 cases, widespread	[cats <1991]	b
Antipodes Island[1]	1990s	Grey-backed storm-petrel *Garrodia nereis*	eggs & (chicks) (<20 g)	extent unknown	none	c
Selvagem Grande Island	1995	White-faced storm-petrel *Pelagodroma marina*	eggs & chicks (<20 g)	frequent	rabbits [<~2008]	d
Gough Island	2000-2001	Tristan albatross *Diomedea dabbenena*	chicks (500 g – 5 kg)	widespread, ongoing	none	e
	2000-2001	Atlantic petrel *Pterodroma incerta*	chicks (40–400 g)	widespread, ongoing	none	e
	2008-2011	Four species of burrowing petrels	eggs & chicks (30–300 g)	widespread, ongoing	none	f
	2000-2007	Tristan albatross *Diomedea dabbenena*	chicks (500 g – 5kg)	widespread, ongoing	none	g
	2003-2008	Atlantic petrel *Pterodroma incerta*	chicks (40–400 g)	6 cases, widespread	none	h
	2003-2008	Great shearwater *Ardenna gravis*	eggs & chicks (30–300 g)	1 case (suspected to be widespread)	none	h
	2008-2010	At. yellow-nosed albatross *T. chlororhynchos*	chicks (500 g – 2 kg)	isolated cases	none	i
	2008-2010	Sooty albatross *Phoebetria fusca*	chicks (500 g – 2 kg)	isolated cases	none	i
Sand Island, Midway[2]	2015	Laysan albatross *Phoebastria immutabilis*	chicks & adults	extensive, ongoing	[rats <1997]	j

[1] mice have recently been successfully eradicated from Antipodes Island (March 2018, https://www.islandconservation.org)

[2] Polynesian *Rattus exulans* & black *R. Rattus* rats were eradicated in 1997, but mice remained

Data sources: (a) Fugler *et al*. 1987; (b) Jones & Ryan 2010; (c) Burger & Gochfeld 1994; (d) Campos & Granadeiro 1999; (e) Cuthbert & Hilton 2004; (f) Cuthbert *et al*. 2013a; (g) Wanless *et al*. 2009; (h) Wanless *et al*. 2007; (i) Cuthbert *et al*. 2013b; (j) Duhr-Schultz pers. comm. 2016.

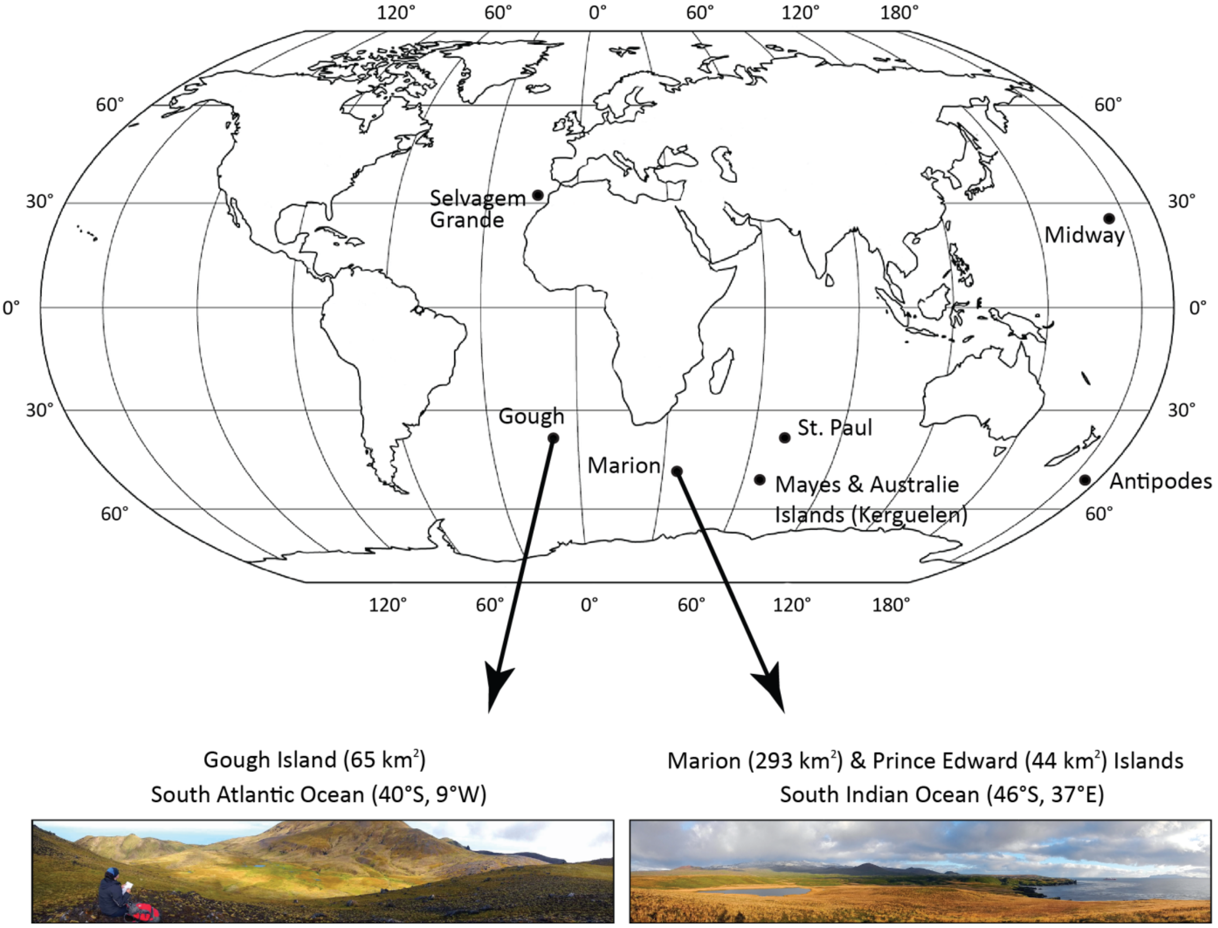


Figure 1.1. The two study islands in relation to six other oceanic islands where seabirds breed which have mice as the sole introduced mammal.

A brief description of the study islands

I completed the fieldwork at two oceanic islands - Marion Island and Gough Island. The two islands are separated by thousands of kilometres of ocean and almost 50° of longitude, but both are globally important breeding sites for hundreds of thousands of seabirds and both have mice as the sole introduced mammal.

Gough Island (65 km^2, 40˚82'S, 9˚85'W, Fig. 1.1, Fig 1.2) lies in the central South Atlantic Ocean and is the southernmost island of the United Kingdom's Overseas Territory of Tristan da Cunha. Gough is a UNESCO World Heritage Site and globally important breeding ground for 23 species of seabird (several of which are globally threatened) and has long been considered one of the world's most important seabird breeding islands (Swales 1965). Gough is the breeding ground for virtually the entire global population of Tristan albatrosses *Diomedea dabbenena* (Critically Endangered (CR); 2–3 pairs also breed on Inaccessible Island); globally important populations of Atlantic yellow-nosed *Thalassarche chlororhynchos* (~25% of global population, Endangered (EN)) and sooty *Phoebetria fusca* albatrosses (~37% of global population, Endangered (EN)); at least 13 species of burrowing petrels, including virtually the entire (99% of global population) population of the Endangered (EN) Atlantic petrel *Pterodroma incerta*. It also hosts significant global populations of grey petrel *Procellaria cinerea* (globally Near Threatened (NT) and regionally Vulnerable (VU)), two shearwaters *Ardenna gravis* and *Puffinus asssimilus* (both Least Concern (LC)), soft-plumaged petrel *Pterodroma mollis* (LC), Kerguelen petrel *Aphrodroma brevirostris* (LC), broad-billed *Pachyptila vittata* (LC), and MacGillivray's prion *P. macgillivrayi* (EN), common diving petrel *Pelecanoides urinatrix* (LC), and at least three species of storm petrel (white-faced *Pelagodroma marina* (LC), grey-backed *Garrodia nereis* (LC) and black-bellied storm *Fregetta tropica* petrels (LC), Brooke 2004a).

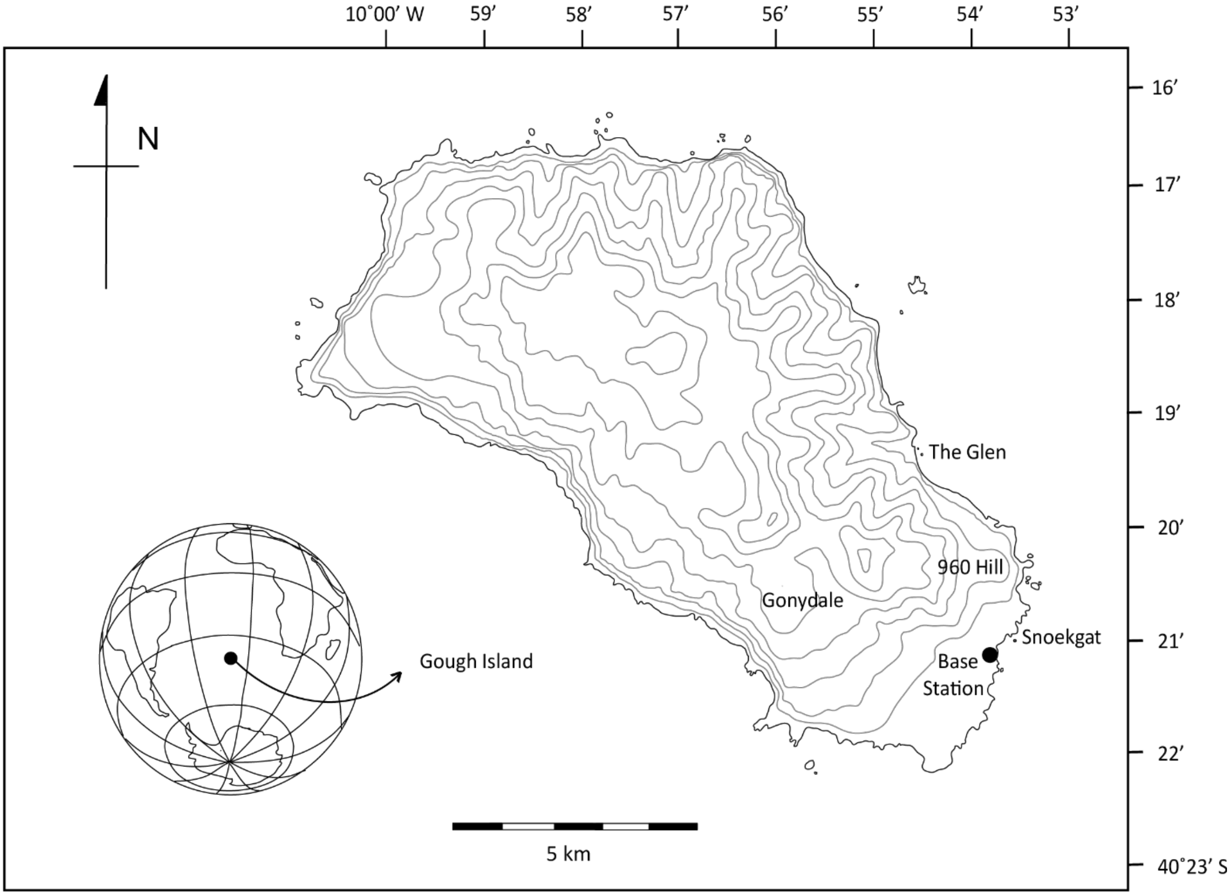


Figure 1.2. A simplified map of Gough Island in the central South Atlantic Ocean, showing the locations of landmarks mentioned in this thesis.

Sub-Antarctic **Marion Island** (293 km^2, 46° 54′S, 37° 45′E, Fig. 1.1, Fig. 1.3) is the larger of the two South African Prince Edward Islands which lie ~2,300 km south-east of Cape Town in the south-western Indian Ocean. As a Special Nature Reserve, established in 1995, the Prince Edward Islands are afforded the highest degree of protection under South African environmental legislation (de Villiers and Cooper 2008). They also have been a Wetland of International Importance (Ramsar Convention since 2007, de Villiers *et al*. 2011) and are surrounded by a large (180,000 km^2) Marine Protected Area declared in 2013 that reaches in places to the edges of South Africa's 200 nautical mile Exclusive Economic Zone around the islands (Lombard *et al*. 2007; Nel and Omardien 2008). A revised management plan adopted in 2014 guides and controls activities at the island group, including biosecurity protocols to avoid alien introductions (DST-NRF Centre of Excellence for Invasion Biology 2014).

The Prince Edward Islands currently support breeding populations of 29 species of birds (Ryan and Bester 2008). Marion Island supports about 25% of the world's breeding population of wandering albatrosses *Diomedea exulans* (globally and regionally (VU)), 12% of the world's breeding population of sooty albatrosses (EN) and 7% of the world's breeding population of grey-headed *Thalassarche chrysostoma* albatrosses (EN) and smaller global percentages of white-chinned petrel *Procellaria aequinoctilais* (VU), grey petrels (globally Near Threatened (NT) and regionally Vulnerable (VU)), and light-mantled albatrosses *Phoebetria palpebrata* (NT).

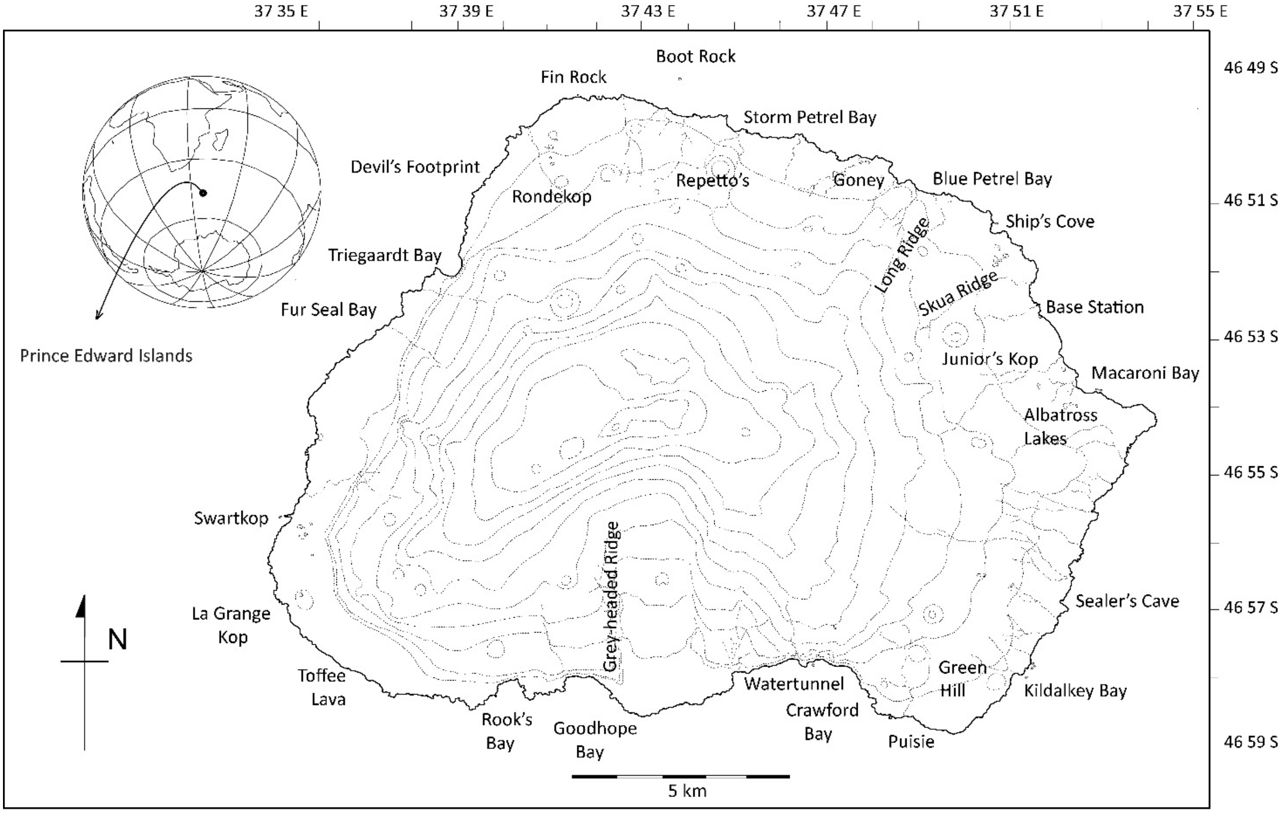


Figure 1.3. A simplified map of Marion Island in the south-western Indian Ocean, showing the locations of landmarks mentioned in this thesis.

Overview of mice depredating seabird chicks at Gough and Marion Islands – a relatively new development

House mice were brought to the Gough Island by sealers in the 19th century and are the only mammal that has been introduced to the island. The impact of mice on Gough Island's seabird populations has received attention since 2001, when mouse predation was identified as the most probable cause of the high chick mortality of Tristan albatross (Cuthbert and Hilton 2004). Further research on Gough in 2004 confirmed at least three species were being depredated by mice (Wanless *et al*. 2005, 2007, 2012; Angel *et al*. 2009). The severe predatory behaviour of mice on Gough has contributed to the Tristan albatross and endemic Gough bunting *Rowettia goughensis* being listed as Critically Endangered (Ryan and Cuthbert 2008; Wanless *et al*. 2009) and it is highly likely that the island's other formerly abundant seabird populations have also declined dramatically because of this (Ryan 2010; Cuthbert *et al*. 2013b).

Mice were accidentally introduced to Marion Island during the sealing era sometime before 1818 and were the sole introduced mammal until 1948, when five domestic cats were introduced to control mice at the newly-established weather station (Watkins and Cooper 1986). However, little was known about the potential harmful effects of invasive species on islands in the 1950s. Rand (1954) was the Biologist on the Eighth South African Expedition to Marion Island over 1951/52 and noted how "a few domestic cats have gone feral and prey on the smaller petrels or mice that are widespread over the coastal plain" (p 178) and "mice often burrow into the [albatross] nest cone but do no appreciable damage" (p 189). Unfortunately, the cats preferred to eat the island's native birds rather than mice, especially the burrow-nesting petrels, and by the 1970s more than 2,000 cats were killing some 450,000 birds each year (van Aarde 1980). As a result, at least one species, the common diving petrel, disappeared from the island and all the other burrowing petrels became far less common than at nearby predator-free Prince Edward Island.

A sustained and multi-faceted eradication programme that commenced in the mid-1970s finally eradicated cats from the island by 1991 (Bester *et al*. 2002), in what is still the largest island area cleared of cats. The first signs of mouse attacks on seabirds at Marion Island were recorded in 2003, when wandering albatross chicks were observed with rump wounds typical of those inflicted by mice on Tristan albatross chicks on Gough Island (Jones and Ryan 2010). The first attacks on summer-breeding albatross chicks at Marion were recorded in April 2009 when sooty albatross fledglings were found 'scalped' with raw, bleeding crowns and necks (Jones and Ryan 2010). In 2015, levels of mouse attacks on large chicks of all three albatross species that fledge in autumn increased sharply and from 2016–2018 attacks on summer-breeding albatross fledglings have continued at varying rates.

Overview of this thesis

This thesis assesses the impacts of invasive mice and establishes pre-eradication baselines for the petrel populations on Marion Island. Although mice predation had been identified as a potentially serious threat to seabirds on Gough and Marion by the early 2000s, little recent progress had been made at either island to monitor how many seabird species are affected and to quantify the impacts. Field observations and anecdotal evidence suggested a marked increase in levels of mouse predation at both islands, yet there was still no direct evidence of mouse depredating burrow-nesting petrels at Marion. In addition, Marion Island has been free of cats since 1991, and an assessment of how burrow-nesting petrel populations have recovered over the subsequent two decades was required. Since starting this fieldwork with a burrow-nesting petrel survey in 2012 at Marion Island, the levels of mouse predation at both islands have reached unprecedented levels, and as such the structure of this thesis evolved to incorporate and quantify these new developments.

The ultimate conservation aim is to aid the restoration of Marion and Gough Islands by eradicating mice from both islands. The scientific and visual evidence collected during this thesis has contributed to the growing body of evidence needed to persuade funders and governments to support the planned eradication campaigns at both islands. Most of the focus in this thesis is on Marion Island because this is where I did most of the fieldwork (overwintering expeditions in 2009–10 and 2012–13 and one relief voyage in 2015), including the island wide surveys of three species of burrowing petrels (Chapter 2, and Appendices 1–2) and establishing burrowing petrel monitoring colonies. The work I did on Gough (overwintering expedition 2013–14 and summer visit in 2015) provided an invaluable contrast to Marion, because although Gough also has mice as the sole introduced mammal, the density of burrowing petrels is much higher (Marion's populations were decimated by cats over four decades) and the level of mouse predation is more severe.

To document and quantify the direct effects of mice attacks on seabird chicks I used an assortment of cameras and systematic colony checks on Marion and Gough, which provided some indication of the levels of mouse predation at localised study sites from 2012–2014. However, it is also important to monitor the long-term changes in the populations of burrowing petrels, especially considering the future plans to eradicate mice from both islands. Both islands lack reliable population estimates for burrowing petrels; on Gough this is largely due to the dense vegetation and rugged topography, which restrict access to much of the island and make systematic island-wide surveys almost impossible. By contrast, Marion Island is relatively flat with very low vegetation, allowing seemingly reliable island-wide surveys to estimate the sizes of the main burrowing petrel populations as benchmarks to assess future population changes.

With these points in mind, I have three broad aims (Fig. 1.4) in this thesis:

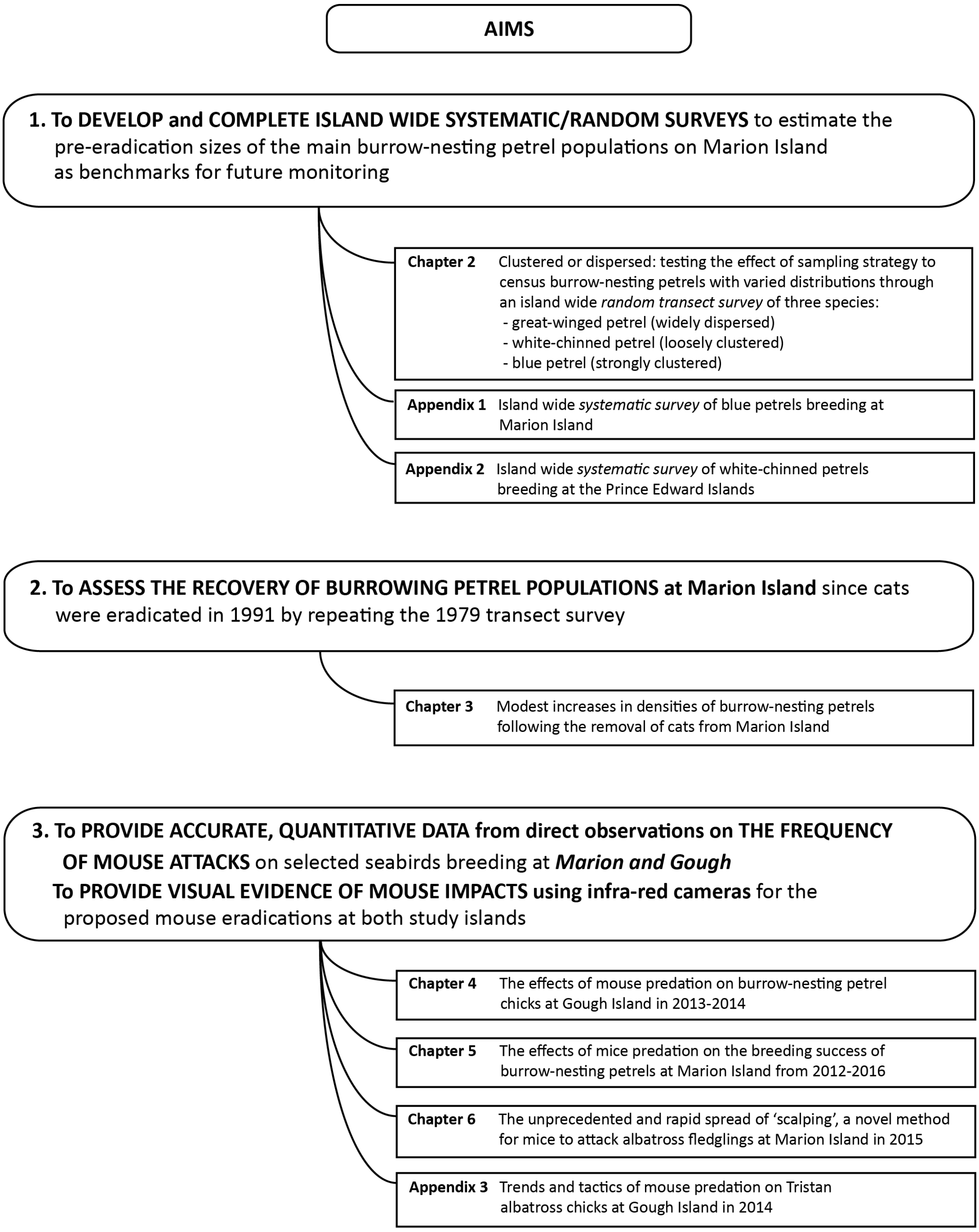
1. To develop and complete island wide systematic/random surveys to estimate the pre-eradication sizes of the main burrow-nesting petrel populations on Marion Island as benchmarks for future monitoring.

2. To assess the recovery of burrowing petrel populations at Marion Island since cats were eradicated in 1991 by repeating the 1979 transect survey.

3. To provide accurate, quantitative data from direct observations on the frequency of mouse attacks on selected seabirds breeding at Marion and Gough.

Thesis outline

The thesis is presented as a series of chapters written as stand-alone documents for submission to journals. As such, there is some repetition in the introduction and methods sections. The three Appendices and four of the five data chapters (Chapters 3–6) have been published in peer reviewed journals. Chapter two has been submitted to *Antarctic Science*. Details of each submission or publication are listed at the start of each Chapter/Appendix.



AIMS

1. **To DEVELOP and COMPLETE ISLAND WIDE SYSTEMATIC/RANDOM SURVEYS** to estimate the pre-eradication sizes of the main burrow-nesting petrel populations on Marion Island as benchmarks for future monitoring

Chapter 2 Clustered or dispersed: testing the effect of sampling strategy to census burrow-nesting petrels with varied distributions through an island wide *random transect survey* of three species:
- great-winged petrel (widely dispersed)
- white-chinned petrel (loosely clustered)
- blue petrel (strongly clustered)

Appendix 1 Island wide *systematic survey* of blue petrels breeding at Marion Island

Appendix 2 Island wide *systematic survey* of white-chinned petrels breeding at the Prince Edward Islands

2. **To ASSESS THE RECOVERY OF BURROWING PETREL POPULATIONS at Marion Island** since cats were eradicated in 1991 by repeating the 1979 transect survey

Chapter 3 Modest increases in densities of burrow-nesting petrels following the removal of cats from Marion Island

3. **To PROVIDE ACCURATE, QUANTITATIVE DATA from direct observations on THE FREQUENCY OF MOUSE ATTACKS** on selected seabirds breeding at ***Marion and Gough***
To PROVIDE VISUAL EVIDENCE OF MOUSE IMPACTS using infra-red cameras for the proposed mouse eradications at both study islands

Chapter 4 The effects of mouse predation on burrow-nesting petrel chicks at Gough Island in 2013-2014

Chapter 5 The effects of mice predation on the breeding success of burrow-nesting petrels at Marion Island from 2012-2016

Chapter 6 The unprecedented and rapid spread of 'scalping', a novel method for mice to attack albatross fledglings at Marion Island in 2015

Appendix 3 Trends and tactics of mouse predation on Tristan albatross chicks at Gough Island in 2014

Figure 1.4. The three aims of this thesis.

In **Chapter 2** I present results from 52 random line transects totalling 145 km on Marion Island to estimate the abundance of great-winged petrel *Pterodroma macroptera* burrows, and to compare the use of random (transects) versus systematic (focussed) searches for blue *Halobaena caerulea* (Appendix 1) and white-chinned petrel *Procellaria aequinoctialis* burrows (Appendix 2). Burrowing petrels are difficult to survey and the development of repeatable, island-specific survey methods is very useful for long term monitoring. Burrow densities were calculated for all three species within the random transects and extrapolated by associated habitat attributes to generate island-wide population estimates. These results showed how random transects appear to be a sensible choice for species that are widely distributed at low densities (e.g. white-chinned and great-winged petrels), but become increasingly poor for estimating species with clustered distributions (e.g. blue petrels), especially on large islands where sampling coverage is going to be a tiny proportion of the island area.

In **Chapter 3**, I repeat Mike Schramm's burrowing petrel survey in the north-eastern sector of Marion Island, which he originally did over the austral summer of 1979/80. The aim of this study was to assess how petrel numbers have changed compared to densities at the peak of the cat-era. The recovery of other summer-breeding species decreased with decreasing body size, and winter-breeding species showed even smaller recoveries, similar to patterns of breeding success at Gough Island.

In **Chapters 4** and **5**, I use infra-red video cameras and burrow-scope nest inspections to monitor the severity and frequency of mouse impacts on multiple species of burrow-nesting petrels at Gough (Chapter 4) and Marion (Chapter 5) Islands. On Gough Island this was done over one year where seven species of burrow-nesting petrels where filmed and monitored. On Marion Island two summer and two winter breeding burrowing petrels species where monitored from 2012–2016 using burrow cameras and nest inspections at study colonies to assess how mice are affecting these species. Results show that mice can be very effective predators of both winter and summer breeding burrowing petrel species at both islands.

In **Chapter 6**, I report on the unprecedented increase in the frequency and distribution of mouse attacks in 2015–2018 on large chicks of all three albatross species that fledge in autumn on Marion Island.

Chapter 7 is a synthesis of the main findings of this thesis, with suggested directions for future research and an update on eradication plans for Marion and Gough islands.

In **Appendix 1** I present results from a systematic whole island survey of blue petrels. These results relate to Chapter 2 where I compare the merits and practicalities of random versus systematic survey techniques to estimate burrowing petrel populations on large islands.

Appendix 2 is a systematic whole island survey of white-chinned petrels. These results relate to Chapter 2 where I compare the merits and practicalities of random versus systematic survey techniques to estimate burrowing petrel populations on large islands.

In **Appendix 3** I present results from a study which Delia Davies and I completed on Gough Island in 2014 where we used infra-red cameras to closely monitor the fates of 20 Tristan albatross *Diomedea dabenena* chicks. We confirmed that albatross chicks can be killed outright by mice which were responsible for 93% of chick failures. Another interesting finding was how quickly the process happens from the initial mouse attack to the death of the chick (average <4 days).

All the work presented in this thesis is my own, although I did have enthusiastic assistance from a number of people in the field over the years. My supervisor Peter Ryan and I worked together on Marion Island to design and carry out the white-chinned petrel survey in 2009 (assisted by Genevieve Jones; Appendix 2); the blue petrel survey in 2012 (assisted by Delia Davies, Mike Schramm and Maëlle Connan; Appendix 1); the great-winged petrel transect survey in 2015 (assisted by Stefan Schoombie, Alexis Osborne and David Hedding (GIS analyses); Chapter 2); and monitoring the spread of mouse attacks on summer-breeding albatross chicks around Marion Island in 2015 (assisted by Stefan Schoombie, Janine Schoombie, Vonica Perold and Alexis Osborne; Chapter 6). Mike Schramm joined us on Marion Island in April-May 2012, where he taught me how to identify different petrel burrows and assisted with the planning of the repeat survey (Chapter 3). Delia Davies assisted with burrow checks and monitoring of burrow cameras on Gough Island in 2013–14 (Chapter 4). On Gough Island, Delia Davies and I both collected the data and wrote the paper on mice predating Tristan albatross chicks in 2014 (Appendix 3). In 2012, I set up study colonies to monitor the breeding success of white-chinned, great-winged and grey petrels on Marion Island which were continued in the subsequent years by Stefan Schoombie, Kim Stevens and Tegan Carpenter-Kling (2013); by Alexis Osborne and Vonica Perold (2014); by Stefan and Janine Schoombie (2015); and by Kim Stevens and Christiaan Brink (2015; Chapter 5).

Chapter 2

Clustered or dispersed: testing the effect of sampling strategy to census burrow-nesting petrels with varied distributions at sub-Antarctic Marion Island

A great-winged petrel *Pterodroma macroptera* guarding its newly hatched chick in a burrow at Marion Island in 2012 (photo Ben Dilley).

This chapter has been *accepted with revisions* (October 2018) for publication by the peer-reviewed journal *Antarctic Science*:

Dilley, B.J., Hedding, D.W., Rexer-Huber, K., Parker, G.C., Schoombie, S., Osborne, A., and Ryan, P.G., submitted. Clustered or dispersed: testing the effect of sampling strategy to census burrow-nesting petrels with varied distributions at sub-Antarctic Marion Island. (submitted June 2018; accepted with revisions October 2018)

Author contributions: BJD, SS, OA & PGR planned the field work at the study site and completed the survey; SS completed the post-survey fieldwork; DWH completed all GIS analyses; BJD analysed the data with assistance from PGR; BJD wrote the complete draft; PGR assisted with manuscript edits/preparation; KR-H & GCP contributed to drafts.

Chapter 2: Clustered or dispersed: testing the effect of sampling strategy to census burrow-nesting petrels with varied distributions at sub-Antarctic Marion Island

Abstract

In this chapter I compare random and systematic surveys to estimate populations of burrow-nesting petrel species on sub-Antarctic Marion Island (293 km^2). Systematic surveys of blue petrels *Halobaena caerulea* (2012, Appendix 1) and white-chinned *Procellaria aequinoctialis* petrels (in 2009, Appendix 2) provided baseline population estimates for these species on Marion. In 2015 I conducted random surveys with 52 randomised strip transects (25 m wide) radiating inland, covering a total of 144 km. I counted all white-chinned, blue and great-winged *Pterodroma macroptera* petrel burrows within transects, and associated five GIS-derived habitat attributes (geology, vegetation, slope, elevation and aspect) with each burrow location. Burrow densities estimated from the random transects were extrapolated to the planar surface area of the five habitat attributes (in 32 combinations) to generate island-wide burrow count estimates. Great-winged petrels were found at 237 sites, at an average burrow density of 2.2 burrow·ha^{-1}. Using the geology-vegetation-elevation model gives an estimate of 32,400 great-winged petrel burrows on Marion (95% CI 15,800–46,300); after correcting for burrow occupancy (42%, 95% CI 29–55%), I estimate a breeding population of 13,700 great-winged petrel pairs in 2015 (95% CI 9,500–17,900 pairs), which is similar to the previous best estimate for Marion Island (10,000 pairs). The random transects found additional breeding sites for white-chinned and blue petrels, accounting for 18% and 4% of the two species' counts, respectively. White-chinned petrels were found at 184 sites at an average of 3.0 burrow·ha^{-1}, and the best extrapolated burrow estimate (49,000, 95% CI 40,200–56,400) was 58% higher than the systematic estimate (30,800). By comparison, blue petrels were found at only 29 sites at an average of 12.9 burrow·ha^{-1}, which extrapolates to 119,500 (102,500–152,800) burrows, 42% less than the systematic estimate of 214,700 burrows. The results suggest that random transects are best suited for species that are widely distributed at low densities, but become increasingly poor for estimating population sizes of species with clustered distributions.

Introduction

Despite being the most abundant seabirds in the Southern Oceans, the global population of burrow-nesting petrels is poorly known (Paleczny *et al*. 2015). Many petrel species breed on remote islands with challenging terrain where burrows are often widely distributed, making their populations difficult to count and monitor accurately (Brooke 2004b). Nearly half of all seabird species are known or suspected to be declining (Croxall *et al*. 2012), with some of the primary causes among burrow-nesting petrels being incidental mortalities in fisheries (Barbraud *et al*. 2009; Nel *et al*. 2002; Petersen *et al*. 2009) and depredation by introduced mammals at their breeding grounds (e.g. domestic cats

Felis catus, rats *Rattus* spp. and house mice *Mus musculus*; Jones *et al*. 2008; Howald *et al*. 2007; Cuthbert *et al*. 2013a, b).

Obtaining reliable estimates of population sizes and trends are central to the effective conservation of seabirds, but collecting quality data for burrow-nesting petrels is often more challenging than for surface nesting birds so burrow-nesting petrels remain relatively understudied (Brooke 2004a). Early records of burrow-nesting petrel numbers on remote oceanic islands were usually crude, order of magnitude estimates of population sizes made from brief visits to breeding colonies (Blackburn *et al*. 2004). More accurate survey methods are increasingly being used and developed to collect quantitative data to estimate population sizes (Rayner *et al*. 2007a; Lormée *et al*. 2012; Schumann *et al*. 2013; Whitehead *et al*. 2014; Appendix 2). These surveys usually involve extrapolation of burrow densities calculated from transects or plots to the area of available nesting habitat within the island or island group (Barbraud and Delord 2006; Lawton *et al*. 2006; Parker *et al*. 2017). The process of extrapolation can result in large errors, since bias (e.g. observer or habitat availability bias) or imprecision (e.g. incomplete detection of burrows or burrow occupants) are also inflated, resulting in inaccurate estimates of population size (Parker and Rexer-Huber 2015). Reducing this error is especially important if estimates are to be used for detecting trends in the population size over time (e.g. short term changes after rodent eradications and long term trends influenced by varied marine threats, climate change and pollution), because small or moderate changes may not be detected if error margins are too large (e.g. Oppel *et al*. 2014). However, error margins around population estimates can be reduced by designing a survey specific to the species and site (Parker and Rexer-Huber 2015).

Sub-Antarctic Marion Island (46°54’S, 37°45’E) in the southwest Indian Ocean provides a sobering example of the consequences of introduced cats and mice, but also an opportunity to develop methodologies to reduce the effects of bias and error when surveying burrow-nesting petrel populations. Mice were brought to Marion Island during the sealing era sometime before 1818, and in 1948 cats were introduced to control mice at the newly-established weather station (van Aarde 1980). The cats soon turned feral, greatly reducing burrow-nesting petrel populations over four decades (Schramm 1986), before cats were finally eradicated by 1991 (Bester *et al*. 2002). In the absence of competition and predation from larger introduced mammal species, and aided by a drier, warmer climate, mice now attain very high population densities in summer (up to 237 mice·ha^{-1}; McClelland *et al*. 2018).

The post-cat recovery of burrow-nesting petrel numbers on Marion has been much slower than anticipated, especially for smaller species (Chapter 3). A repeat survey of burrow densities (Chapter 3) and analyses of brown skua *Catharacta antarctica* prey remains (Cerfonteyn and Ryan 2016) both suggest there has been at best a limited recovery of burrow-nesting petrel populations at Marion since cats were eradicated. There is compelling evidence that the continuing mice predation impacts are influencing the recovery of Marion’s petrel populations (Chapter 3). Recent breeding success studies show that mice are suppressing the recovery of burrow-nesting petrel populations, especially petrel species that breed in winter when mice are more desperate for food and depredate petrel eggs and chicks (Chapters 4 and 5).

A mouse eradication attempt using aerial baiting is planned for the winter of 2021 (Preston *et al*. 2018). In preparation for this, I endeavoured to establish pre-eradication baseline population estimates of three key burrow-nesting petrel species. The distribution and abundance of blue petrels *Halobaena caerulea* and white-chinned petrels *Procellaria aequinoctialis* were assessed with independent systematic surveys at Marion Island in 2012 (Appendix 1) and 2009 (Appendix 2), respectively. In this chapter I test the effect of sampling strategy (random transect or systematic survey) on population size estimates of burrow-nesting petrels on a large sub-Antarctic island. White-chinned petrel, blue petrel and great-winged petrel *Pterodroma macroptera* burrows were recorded during random 25 m wide transects conducted from the coast inland to the upper limit of petrel breeding habitat. I hypothesise that random transects will be suitable for species such as great-winged petrels that are widely distributed at low densities, but become increasingly poor for estimating species with loosely clustered (white-chinned petrels) or strongly clustered distributions (blue petrels). I also provide the first robust population estimate of breeding great-winged petrels for Marion Island.

Methods

Study species

The great-winged petrel is a large gadfly petrel (~500–650 g) which breeds on at least six remote oceanic island groups within the Southern Oceans. The global population estimate of some 1.5 million individuals is based on rough estimates at known breeding localities (Brooke 2004a), but accurate estimates are lacking, especially for the (presumed) major breeding sites at islands within the Kerguelen Archipelago and at the Prince Edward Islands. Historically this species was harvested for food by islanders at Tristan da Cunha (Elliot 1953; practice largely ceased since late 1950s), Amsterdam and Saint-Paul Islands (Micol and Jouventin 1995; populations now reduced to a few pairs on these islands). As a winter breeder (Fig. 2.1) their chicks are especially vulnerable to depredation by introduced predators such as cats (van Aarde 1980) and mice (Chapter 5). Although the global population is suspected to be in decline, the species is evaluated as Least Concern (globally) and Near Threatened (regionally) due to its extremely large range and its population size (BirdLife International 2018).

As summer breeders, white-chinned petrels (~950–1,800 g) and blue petrels (~160–250 g) are less affected by mice depredating their eggs and chicks than winter breeding species (Chapter 5). Habitat preferences of blue petrels and white-chinned petrels are detailed in Appendices 1 and 2.

Study area

Marion Island is the summit of a shield volcano and is relatively flat for its size (293 km^2), with a predominantly gentle slope rising to 1 240 m in the central highland (Hedding 2008). The geology is dominated by older Pleistocene basaltic ‘grey’ lavas and younger Holocene ‘black’ basaltic lavas and scoria (Boelhouwers *et al*. 2008). Above 750 m, the island is dominated by a barren polar desert biome (Hedding 2008). Areas below 300 m are well vegetated, especially the steep vegetated slopes of the grey lava ridges, which have well-drained soils and support mosaics of tussock grassland, herbfield and fernbrake (Huntley 1971). Several volcanic eruptions subsequent to glaciations have resulted in extensive areas of broken rocky black lava flows and scoria cones which now cover 80% of Marion

Island. The older black lavas are now undulating vegetated hummocks, whereas the younger flows remain raw black lava expanses with little to no soil or vegetation cover. More detailed descriptions of the geological and vegetation classes used are listed in Appendix 2.1.

Designing the survey

My supervisor Peter Ryan and I led systematic island-wide surveys of white-chinned (16 April to 5 May 2009, Appendix 2) and blue petrels (18 April to 6 May 2012, Appendix 1) on Marion Island. In these surveys, I systematically visited all suitable-looking habitat (also known as a 'targeted survey' approach whereby all known colonies and all suitable habitat are visited on the ground and surveyed), paying particular attention to areas with the tussock grass *Poa cookii*, which is indicative of seabird manuring (Smith 1976; Schramm 1986; see methods in Appendix 1 and Appendix 2). Systematic surveys were chosen since both species have clustered distributions: white-chinned petrels breed in loose colonies and blue petrels in dense colonies. From my field observations during these surveys, I noted great-winged petrel burrows were widely dispersed at low densities. To test the accuracy of random and systematic survey methods for estimating populations of petrels with varying spatial distributions, I conducted random strip transects. I selected 52 random transect lines by starting where the coast intersected each one degree (1') of longitude and half a degree (½') of latitude, and proceeded inland perpendicular to the coast to the upper limit of suitable habitat Figs. 2.2–2.5). The burrows of my study species are not found in bare-rock/unvegetated habitat on Marion Island, so transect lines ended at the vegetation line (250–400 m elevation). Start and end waypoints of transects were determined in Google Earth prior to walking transects and located using hand held Garmin Global Positioning System (GPS) units in the field. Initially I tried sampling burrows within 25 m^2 circular plots every 50 m along the transect line, but this type of random sampling proved ineffective to represent the widely dispersed great-winged petrel burrows and even less effective for the more clustered blue and white-chinned petrel burrows. I settled on a 25 m wide strip along the full length of the line transect, where two observers walked in parallel on the outer edges of the strip (25 m apart as this was the most practical way to define the edge) and counted all burrows within the strip transect. To minimise observer bias, all four observers (myself, Peter Ryan, Alexis Osborne and Stefan Schoombie) walked the first transect together to standardise the methodology and identification of burrows; thereafter, we walked in pairs, regularly switching partners. Strip transects were later broken down into smaller plots of a standard size (25 m wide x 50 m long), which resulted in a string of adjacent plots extending inland from the coast.

Identifying burrows

Two pairs of people completed 2–6 transects per day, and all 52 transects were walked between 14 April and 1 May 2015. This coincides with the annual relief voyage to Marion Island, which is the only time when there is a sufficiently large team on the island to undertake such a labour intensive task. This was before the great-winged petrels lay (laying occurs from late May to early July, Fig. 2.1), but breeding pairs were renovating their burrows before their pre-laying exodus. Great-winged petrels have some overlap with white-chinned petrels which breed in summer (Fig. 2.1), however most white-chinned petrel chicks have already fledged or are near to leaving their burrows by late April. At this time of year, white-chinned petrel burrows could be identified by tufts of down stuck to vegetation at the burrow entrance (from near-fledged chicks exercising their wings at night), an acrid smell of a well-used burrow, and no fresh vegetation lining. Although this survey was outside of the blue petrel's summer breeding season, blue petrels were present as they return to the island from mid-April to mid-May, after a post-breeding moult period at sea (Fugler *et al.* 1987), to re-occupy and renovate burrows.

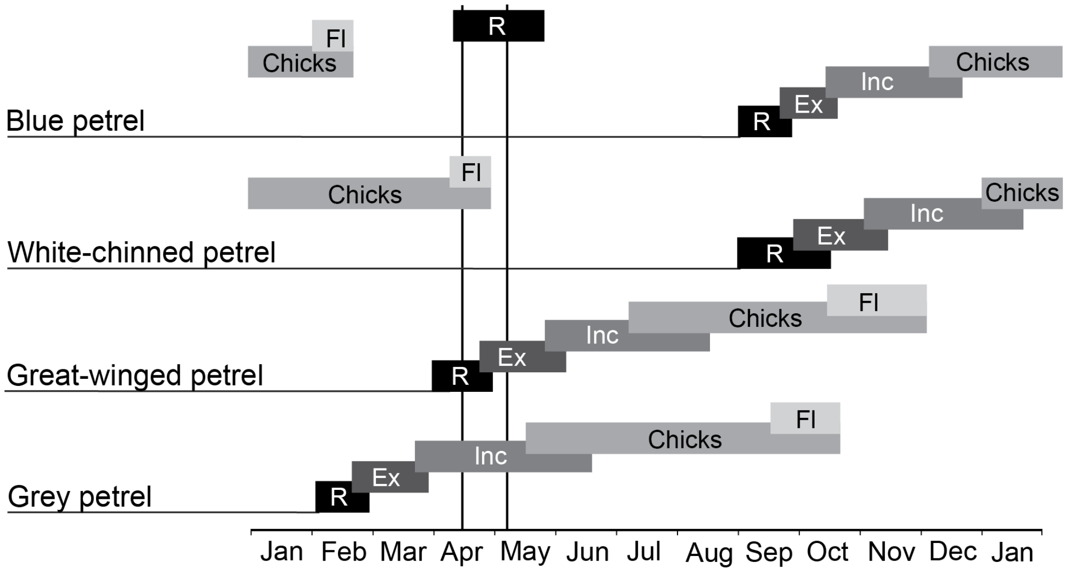


Figure 2.1. Breeding cycles of four species of burrow-nesting petrels on Marion Island. Abbreviates refer to the periods of renovation of burrows (R), pre-lay exodus (Ex), laying and incubation (Inc), hatching and chick rearing (Chicks) and fledging (Fl). The period between the vertical lines indicates the timing of the white-chinned (2009), blue (2012) and great-winged petrel (2015) surveys during the annual relief voyages. Although not surveyed, the grey petrel breeding cycle is included here for comparison with great-winged and white-chinned petrels which also have large burrows.

Petrel burrows were located and identified using a combination of these indicators:

Vegetation: Pockets of taller vegetation where there is deep soil - mainly *Blechnum penna-marina* or *Acaena magellanica*/*Poa cookii* mix. Single burrows often have a few tufts of *Poa* at the entrance (Smith 1976) which is easily spotted on a *Blechnum* slope, even at a distance.

Burrow entrance characteristics: Active burrows showed signs of fresh excavations, feathers, down and fresh guano; active great-winged petrel burrows have extensive cropping of vegetation adjacent to the burrow entrance in a distinct 'oblong' L shaped strip and typically have a dry tunnel entrance, whereas white-chinned petrels crop vegetation randomly around the burrow entrance and usually have a moat or wet mud in the entrance tunnel (Chapter 3); blue petrels tend to have very sparse nest lining and consequently very little cropped vegetation around the entrance.

Burrow entrance shape and size: Great-winged petrel burrow entrances are generally slightly smaller and lower (average 217 ± 50 mm wide by 167 ± 33 mm high) than white-chinned (269 ± 28 mm by 207 ± 31 mm) and grey petrel (238 ± 32 mm wide by 200 ± 29 mm) burrow entrances (Chapter 3). Blue petrel entrances are markedly smaller (142 ± 23 mm wide by 110 ± 10 mm), but readily distinguished from the even-smaller burrows of Salvin's prions (112 ± 10 mm wide by 96 ± 9 mm, Appendix 1).

Visual or physical confirmation: Observing a bird in the burrow or feeling a response to probing into the nesting chamber of the burrow with a stick (Appendix 2).

Grey petrels *P. cinerea* are the only other petrel species on Marion which also breed in large burrows (Schramm 1986). Grey petrels breed in caves and burrows on Marion (Schramm 1986; Chapters 3 and 5), but nests are extremely scarce. Grey petrel burrows are renovated in late February with peak laying from late March to mid-April (Fig. 2.1; FitzPatrick Inst. unpubl. data), so at the time of the survey grey petrels were incubating. Observations from a recent study indicated some burrows are shared between summer and winter-breeding species on Marion Island (Dilley *et al.* in press), where white-chinned petrels evicted and killed 3% of great-winged petrel chicks to claim the burrow. I was unable to quantify how frequently petrels share burrows seasonally, but some recently vacated white-chinned petrel burrows might be used by great-winged petrels (and vice versa).

Only burrows which were overgrown or obviously collapsed (judged by the entrance; burrows with an intact entrance but collapsed chamber were not quantified) were excluded from the count; all other burrows were identified using the indicators above and included in the count. I did not account for burrow detection probability since the low vegetation allowed observers to easily detect burrows in the 25 m wide strip transect. For further details of how blue and white-chinned petrel burrows were identified during the systematic surveys, please see methods in Appendix 1 and Appendix 2.

Estimating the number of burrows

Burrow counts and locations

When a burrow was found, I recorded the species, aspect and the distance from the coastal start point (using a GPS, marked when accuracy <3 m). I estimated the location of a burrow within a transect strip as longitude, calculated as:

$$\mathrm{L} = \mathrm{c}\ \pm \left(\frac{\mathrm{d}}{\mathrm{t}} \times (\mathrm{c} - \mathrm{e})\right)$$

where L = latitude or longitude of a burrow on the transect in decimal degrees; c = coastal start point of the transect, longitude or latitude; d = distance in meters from the burrow to the coastal start point (c); t = total distance of the transect in meters; e = inland end point of the transect. Note that the calculation is added to or subtracted from the coastal start point (c) according to the transect orientation.

Associating habitat variables to each burrow and elevation cut-offs

I selected five habitat attributes most likely to affect the distribution and abundance of petrel burrows: geology (G), vegetation (V), slope (S), elevation (E) and aspect (A). These habitat attributes were selected based on the habitat preferences of burrow-nesting petrels reported in Schramm (1986) and on my own field experience. Spatial data of the five habitat attributes were plotted using the Geographic Information System ArcGIS® 10.1. Slope and aspect data were derived from a 5 m Digital Elevation Model generated using 10 m contour data for Marion Island supplied by National Geo-spatial Information (NGI), a component of the Department of Rural Development and Land Reform (DRDLR) in South Africa. Slope data were categorised into 5-degree segments (e.g. 0–5; 5–10) up to 45 degrees with the last category being 45–90 degrees. Aspect data were divided into eight categories of equal extent of 45 degrees (e.g. N; NE; E). Geology and vegetation spatial data were obtained from Boelhouwers *et al*. (2008) and Mucina and Rutherford (2006), respectively. The spatial locations of burrows for each target species were mapped within each 25 m wide transect using hand held GPS units with a horizontal accuracy of approximately three meters. The five habitat attributes associated with each burrow location could then be determined. I chose the elevation cut-offs (rounded up to the nearest 50 m) for density extrapolations based on the highest burrow recorded for each species in this transect survey: great-winged petrels 350 m, white-chinned petrels 450 m and blue petrels 200 m. In addition, mapping of burrow locations in relation to habitat attributes demonstrates that some habitat attributes (e.g. grey lava, recent black lava flows, scoria cones) are not suitable for burrowing by the target species. The methodology used excluded unsuitable areas because there were no burrows detected in these habitats. Burrow densities were calculated only for habitat attributes where burrows were found.

Burrow densities, extrapolation to estimate the number of burrows and confidence intervals

Using GIS, surface areas of the various habitat attributes (and their sub-categories) were calculated within the 52 transects and within the total island area being sampled (species specific). Burrow densities within transects were calculated as the number of burrows counted in each attribute relative to the attributes surface area. For example, for the model combining geology-vegetation-elevation (GVE), a total of 15 great-winged petrel burrows (sum for all 52 transects) fell within the sub-category of "(Eastern Succession) – (Fellfield) – (100–150 m)", and I estimated the number of burrows for this combination as:

$$\mathrm{B} = \left(\frac{\mathrm{n}}{\mathrm{a}}\right) \times \mathrm{A}$$

where B = estimated number of burrows across the island for this sub-category; *n* = total number of burrows counted within the 52 transects with these three associated attributes; a = total surface area of these three attributes within the 52 transects; A = total surface area of these three attributes within the total island area being sampled (species specific; here below 350 m for great-winged petrels). The island estimates were calculated as the sum of the estimated number of burrows for the various sub-categories across the island. All area calculations were based on planimetric area using a Transverse Mercator projection.

To estimate confidence intervals for the extrapolated population estimates, I used burrow counts in standardised plots generated within the strip transects (plots were 25 m wide x 50 m length, extending inland from the coast) with no associated habitat variables. These plot count data were grouped by 50 m elevation bands. I dealt with the excess of zeros (plots with no burrows) by hierarchical non-parametric bootstrapping (run using library boot (Canty and Ripley 2012) in R (R Core Team 2014) with 5,000 iterations (Chin 2010) of the data to generate means, standard errors and 95% confidence intervals for each elevation band. The confidence intervals were applied to the extrapolated population estimates by elevation band.

Comparison with systematic surveys

For comparison with the systematic survey results I used the estimated number of burrows (before adjusting for occupancy). Burrow estimates are reported in the text to the nearest 100 burrows (original data in Appendices 2.2–2.5) and means ± SD unless otherwise stated. For comparison of the survey effort between systematic and random surveys, I compared the 'main survey person days' (i.e. total person days to complete main survey during the relief voyages), the 'finish-off days' (i.e. total survey days I spent completing the survey after the relief voyages) and the occupancy trial days (for great-winged petrels I had 10 one-off occupancy sites and three checks of 100 burrows near the Research Base; for white-chinned petrels I had 15 one-off occupancy sites and three checks of 100 burrows near the Research Base; and for blue petrels I had 30 one-off occupancy sites and three checks of 200 burrows (100 burrows at two sites) near the Research Base.

Burrow occupancy and estimating the number of breeding pairs

Rates of great-winged petrel nest occupancy were checked by Stefan Schoombie from 9 June to 7 July 2015 (early incubation) at 10 sites around the island which represented a range of habitat types used by great-winged petrels. At each site 20–40 burrows were identified using the same indicators as during the survey (i.e. only burrows which were overgrown or obviously collapsed were excluded) and each burrow was assessed for occupancy on only one occasion. At each burrow, the burrow entrance was inspected and scored as recently active (showing fresh signs of activity, see burrow entrance characteristics above) or inactive; a call playback was then played for 20–30 seconds in the burrow entrance and a response/no response recorded; and lastly, if no response was obtained, a 1.5 m x 25 mm wooden rod was inserted as far as possible to try and feel if a bird was present (grubbing). In addition to these 10 sites, 100 burrows close to the base station were identified to quantify the proportion of active-looking burrows which contained incubating petrels (*% of active burrows occupied by incubators*). All marked burrows were carefully checked on three occasions from 4–13 July 2015 using a burrow-scope (custom-made burrow-scope with a high resolution conical pinhole camera, LED torch and an 18x21 cm colour monitor) to confirm burrow occupancy status. Since great-winged petrel burrows are fairly large and relatively straight, burrows could be fully inspected with the burrow-scope and I considered it unnecessary to quantify occupant detection rates. The very few burrows which were too deep to inspect with confidence were excluded from this occupancy trial. The number of breeding pairs of great-winged petrels was calculated as:

$$\text{breeding pairs} = \left(\frac{\text{a} \times \text{o}}{\text{n}}\right) \times (\text{GVE})$$

where breeding pairs = estimated number of breeding pairs of great-winged petrels on Marion Island in 2015; a = sum of active-looking burrows from all 11 sites, i.e. from the (305 burrows at 10 trial sites around the island) + (100 burrows checked multiple times at the site near the base); o = occupancy of active-looking burrows (where occupancy was confirmed with a burrow-scope at the 100 burrows checked multiple times at the site near the base); *n* = number of burrows sampled at the 11 trial sites (405 burrows), counted in the same manner as burrows in the transects around the island; GVE = estimated number of great-winged petrel burrows around the island using the geology-vegetation-elevation model.

Burrow occupancy and estimating the number of breeding pairs

I used Ivlev's (1961) selectivity index to visualise the extent of burrow-nesting habitats utilised by great-winged and white-chinned petrels relative to the habitat abundance within in the sampled area. Ivlev’s values scale from -1 to 1, where -1 indicates total avoidance of the habitat type, 0 indicates that habitat is utilised in proportion to its relative abundance in the sampled area and 1 indicates a total preference for a habitat type. I calculated Ivlev’s selectivity values as:

$$\text{Ivlev}'\text{selectivity index} = \left(\frac{P_{burrows} - P_{habitat\ area}}{P_{burrows} + P_{habitat\ area}}\right)$$

Where $P_{burrows}$ is the relative abundance of burrows within a specified habitat and $P_{habitat\ area}$ is the relative abundance of the specified habitat within the area of island sampled.

Results

A total of 143.9 km of transect were surveyed (52 transects, average 2.7 ± 1.2 km long, range 0.9–6.7 km), representing a total sampled area of 360 ha. Despite spending 48 person-days on the survey, this represents only 1.2% of the island area (2.0% below 350 m). The relatively flat terrain allowed access by foot to all transects apart from two inaccessible areas of steep, barren cliffs near the coast at Crawford Bay (Figs. 2.2–2.5, each <5% of the respective transects); I assumed that these rocky habitats were unlikely to support any burrows.

Great-winged petrels

Great-winged petrel burrows were recorded in 44 of the 52 transects around the island, most frequently in transects on the north and north-eastern sections of the island (Fig. 2.2). Burrows were not found in sections where transects intersected large black lava flows (e.g. Toffee Lava, La Grange, north of Kaalkop, north of Triegaardt Bay) where the soils are too shallow to support burrows. Some other transects which intersected patches of seemingly favourable habitat also did not include any

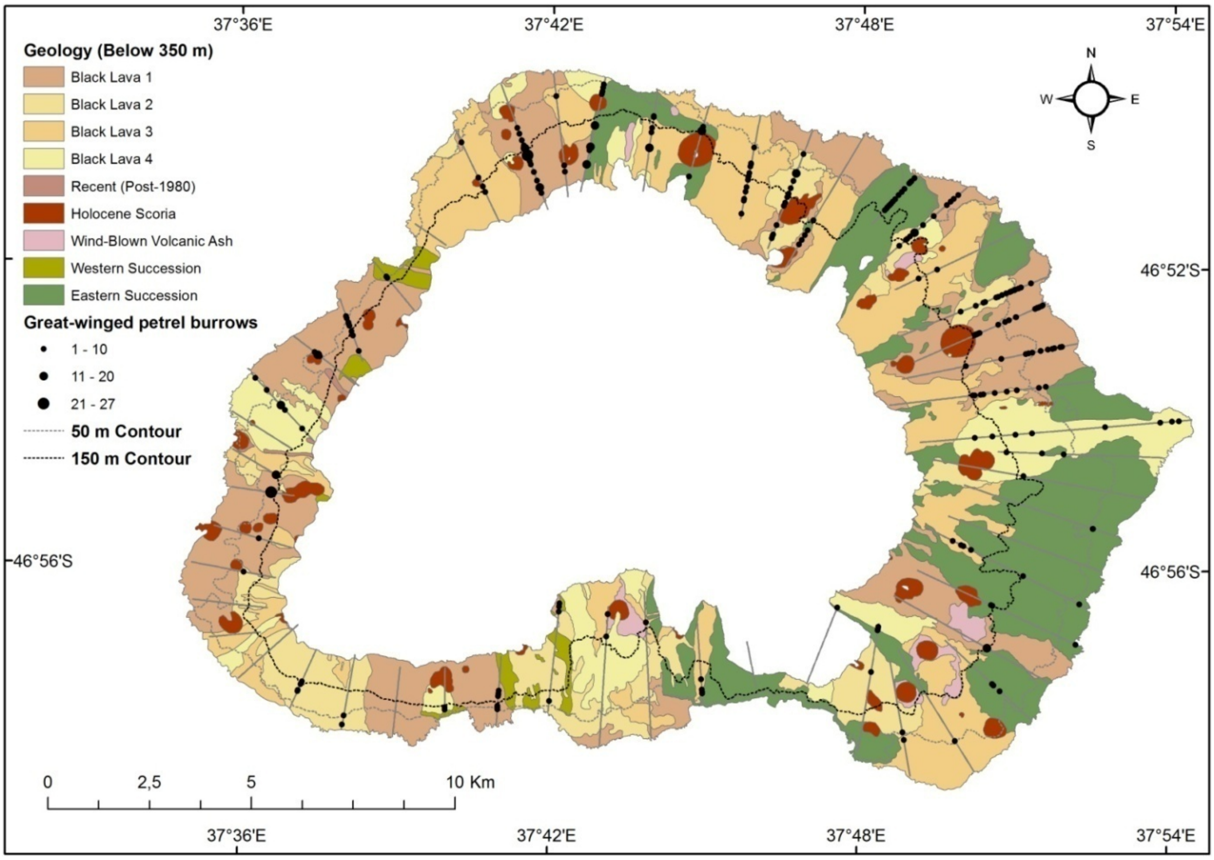


Figure 2.2. The distribution of great-winged petrel burrows found along 52 random strip transects (25 m wide, lines running inland from the coast) sampled at Marion Island in 2015 relative to the main geological features (below 330 m) (Boelhouwers *et al*. 2008).

burrows (e.g. the east coast inland from Killerwhale Cove). A total of 774 burrows was recorded within transects at 237 sites (Table 2.1), with most sites having only one (35%) or two (19%) burrows. Great-winged petrel burrows were found up to 5.3 km inland from the coast at East Cape and up to 325 m elevation inland of Repetto's Hill (Fig. 2.2), with the greatest concentrations of burrows in the northern sector of the island (51% of all burrows on transects between Devil's Footprint and Ship's Cove).

Table 2.1. The estimated number of great-winged, white-chinned and blue petrel burrows (before correcting for occupancy) on Marion Island using systematic (whole island counts) and random (52 transects 25 m wide, total 144 km) sampling techniques. The % difference is relative to the relevant systematic estimate.

	Great-winged petrel	**White chinned petrel**	**Blue petrel**
Summary of random sampling			
Total burrow count in 52 transects	774	1,070	2,655
Extrapolation capped at elevation	350 m	450 m	200 m
Island area sampled	16,030 ha	18,982 ha	9,017 ha
Transects area sampled (% of island area)	351 ha (2.2%)	360 ha (1.9%)	205 ha (2.3%)
Burrow density (burrows/ha in transects)	2.2	3.0	12.9
Number of sites with burrows	237	184	29
Average ± SD burrows per site (range)	3.3 ± 3.5 (1–27)	5.8 ± 7.2 (1–46)	92.4 ± 108.5 (5–500)
Transects with no burrows	8 (15%)	8 (15%)	39 (75%)
***Estimated number of burrows*[1]**			
Best estimate	32,400	48,500	124,100
95% CI of best estimate mean[2]	15,800–46,300	20,700–71,100	40,500–191,000
Range of all 32 estimates	28,500–36,800	40,200–56,400	102,500–152,800
Median of all 32 estimates	33,500	49,000	116,200
Comparison with systematic survey results			
Systematic survey burrow estimate	-	30,800[3]	214,700[4]
Null model (% difference)	35,300	56,400 (+83%)	116,300 (-46%)
Average 32 models (% difference)	33,600	49,000 (+59%)	119,500 (-44%)
GVE model (% difference)	32,400	48,500 (+58%)	-
GVEA model (% difference)	-	-	124,100 (-42%)

[1] Burrow densities estimated from 52 random transects and extrapolated to the planar surface area of five habitat attributes (geology, vegetation, slope, elevation and aspect in 32 model combinations) to generate island-wide estimates; GVE = model based on *geology-vegetation-elevation*; GVEA = *geology-vegetation-elevation-aspect*; [2] count data were bootstrapped with 5,000 iterations (see methods for details); [3] Appendix 2; [4] Appendix 1.

Extrapolation to the whole island (<350 m) with the null model (i.e. extrapolation without using any explanatory environmental variables) estimated 35,300 great-winged petrel burrows. Estimates from 32 model combinations ranged from 28,500 burrows (model GVEAS combining the habitat attributes *geology-vegetation-elevation-aspect-slope*) to 36,800 (model AS combining the habitat attributes *aspect-slope,* Table 2.1, Appendix 2.2). The model combining the habitat attributes *geology-vegetation-elevation* (GVE, Appendix 2.3) appeared to best represent the distribution of great-winged petrels, in particular the high frequency of burrows from 50–150 m (51% of the estimated 32,400 burrows, Fig. 2.3) where birds favour the deep soils associated with the oldest black lavas in the Fellfield and Mire-slope habitats (44% of burrows, Appendix 2.3).

Great-winged petrel burrow entrances predominantly faced north (8%), north-east (48%) or east (20%; from field data), however *aspect* was not a good extrapolation attribute to represent the preferred landscape since the orientation of burrow entrances is not necessarily the same as the overall slope aspect, given options for micro-habitat selection. For example, a south-east facing slope in undulating vegetated lava hummocks may have great-winged petrel burrows in both north-east and south-west facing 'micro' slopes in small gullies. Similarly, based on GIS data *slope* was not an especially good indicator of great-winged petrel preferred habitat; although birds mostly avoided steep slopes (only 18% of burrows were on slopes >20°), there was an even distribution for the remaining burrows on slopes of 0°–20°. Most of the preferred habitats appear to be underutilised in proportion to their relative abundance (Fig. 2.4), with only the more weathered 'black lava 1' habitats showing positive Ivlev selectivity values (Fig. 2.4). Using the GVE model results, I estimate there were 32,400 great-winged petrel burrows on Marion Island in 2015 (95% CI 15,800–46,300).

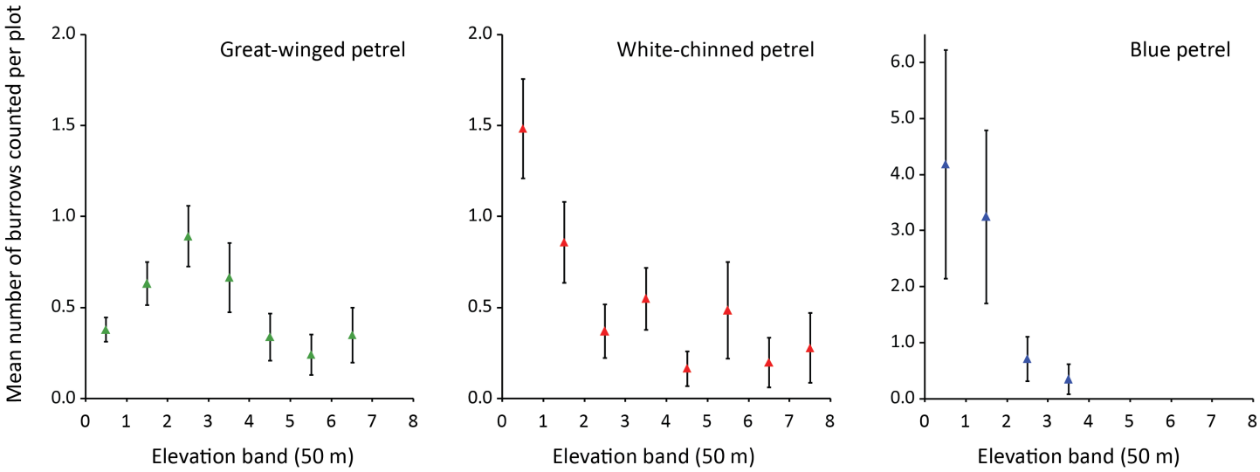


Figure 2.3. The mean (± SE) number of blue, white-chinned and great-winged petrel burrows counted within sample plots (25 m wide x 50 m length) along 52 random transect lines extending inland from the coast at Marion Island. Count data were bootstrapped by elevation band (hierarchical non-parametric bootstrap analysis, run using library boot (Canty and Ripley 2012) in R (R Core Team 2014) with 5,000 iterations) to generate the means and standard errors presented here. Note the y-axis scale is different for the blue petrels.

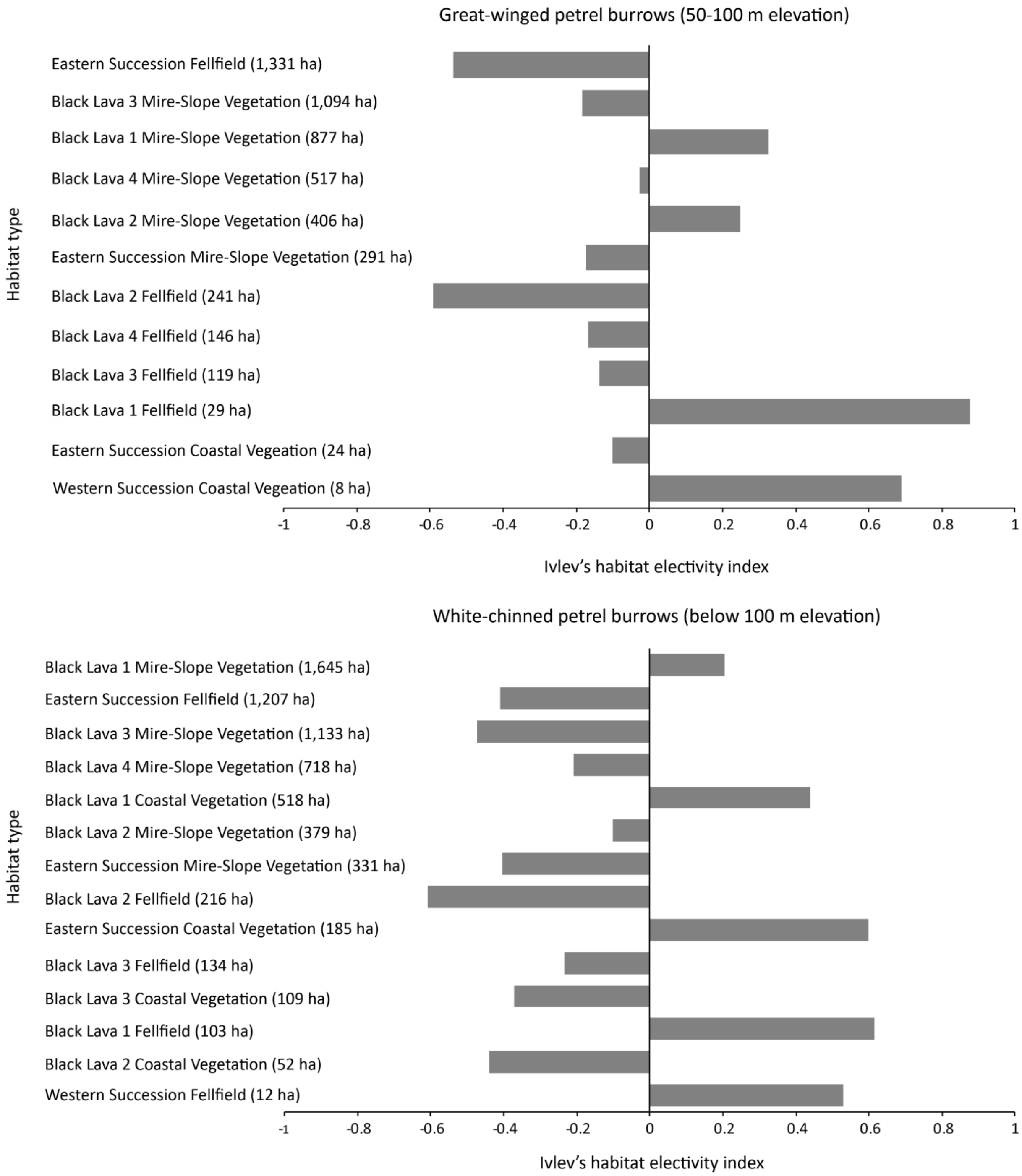


Figure 2.4. The extent of burrow-nesting habitats utilised by great-winged and white-chinned petrels breeding within the specified altitudes at Marion Island. The proportions displayed are Ivlev's (1961) selectivity index values, where -1 indicates total avoidance of the habitat type; 0 indicates that habitat is utilised in proportion to its relative abundance in the sampled area (hectares in parenthesis, listed in descending order); and 1 indicates a total preference for a habitat type. Burrow density was calculated from counts within 52 random strip transects (25 m wide, lines running inland from the coast) sampled at Marion Island in 2015 relative to the main geological features.

One-off occupancy checks of 305 great-winged petrel nests (31 ± 8 nests per site) showed that the proportion of nests with signs of recent activity varied considerably (average 48 ± 29%, range 10–94%) between the 10 sites around the island. Birds were reluctant to respond to call playbacks at active-looking nests (3% response rate to call playback), and grubbing proved impractical to confirm occupancy since many burrows were too deep to confirm an occupant. Of the 100 marked nests visited three times, only 3 burrows were too deep to see the chamber and thus excluded from occupancy analyses. Of the remaining burrows, all 17 with no signs of recent activity were unoccupied. Eighty showed signs of recent activity ('active-looking') on multiple checks. Of these burrows, 14 were unoccupied, 7 contained loafing birds and 59 contained incubating petrels. This suggests that 74% of active-looking burrows were occupied by breeding pairs. Applying this occupancy rate to the proportion of active looking burrows for all burrows sampled suggests an overall occupancy of burrows counted in transects of 42% (95% CI 29–55%). Using this occupancy, the best-estimate of 32,400 burrows and a 95% CI of 0.42–0.63 (estimated from the mean great-winged petrel burrow count per plot (25 m x 50 m) of 0.53 ± 0.05 (SE)), suggests a great-winged petrel breeding population estimate of 13,700 pairs on Marion Island in 2015 (95% CI 9,500–17,900).

Comparison with systematic white-chinned and blue petrel surveys

A total of 1,070 white-chinned petrel burrows was recorded from 184 sites along 44 of the 52 transects (Table 2.1), with the greatest concentrations in the north-west (Swartkop to Fur Seal Bay, 35% of all burrows) and the north-east (Repetto's Hill to Sealers' Beach, 19%) sections of the island (Fig. 2.5). Most sites had moderate groupings of burrows (5.8 ± 7.2 burrows per site), but 51% of sites contained only 1–3 burrows. Three moderate sized colonies were found which were missed in the systematic survey in 2009: inland of Cape Hooker on the south-east coast (20 burrows), south of Kaalkoppie (18) and inland of Fur Seal Bay (35) on the west coast (Fig. 2.6). An additional 39 previously unrecorded sites were found around the island (Fig. 2.6), mostly inland of the high density coastal areas (200 missed burrows overall, 18% of total count). Estimates from 32 model combinations (<450 m) ranged from 40,200 burrows (GVEAS) to 56,400 burrows (Null model) (Table 2.1, Appendix 2.2). These estimates are 31–83% greater than the systematic estimate of 30,800 burrows (Appendix 2) (Table 2.1). The combination of the habitat attributes *geology-vegetation-elevation* (GVE, Appendix 2.4) appeared to best represent the habitat preferences of white-chinned petrels based on my field experience and understanding of the petrels habitat preferences around Marion. For example, in this model 78% of burrows occurred below 100 m elevation and 55% of white-chinned petrel burrows occurred in

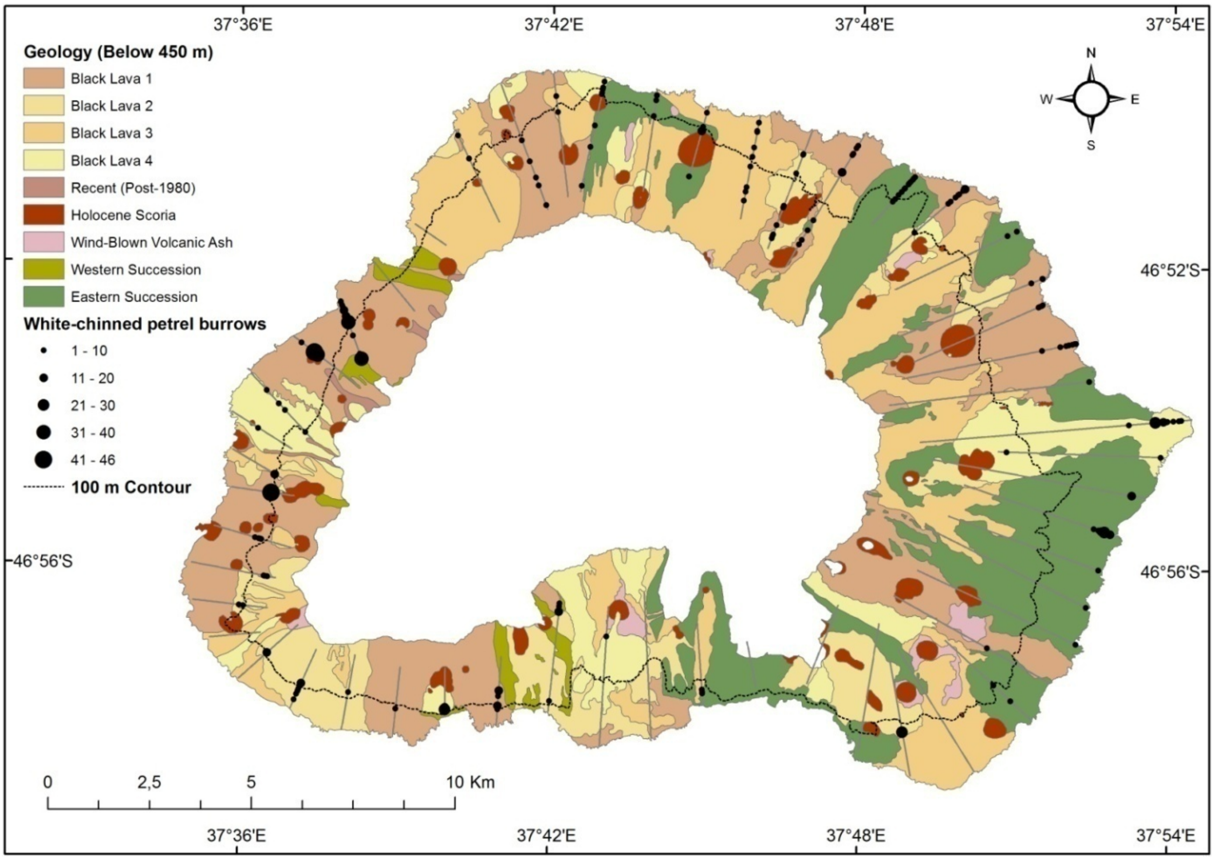


Figure 2.5. The locations of white-chinned petrel burrows found along 52 random strip transects (25 m wide, lines running inland from the coast) sampled at Marion Island in 2015 relative to the main geological features (below 330 m) (Boelhouwers *et al*. 2008).

either the Coastal, Fellfield or Mire-Slope vegetation on the oldest (most weathered) 'black lava 1' habitats (see Fig. 2.4 where habitats with the geological attribute 'black lava 1' had the highest Ivlev's selectivity values). Using the GVE model I estimated there were 48,500 white-chinned petrel burrows on Marion Island in 2015 (95% CI 20,700–71,100; CI based on mean white-chinned petrel burrow counts per plot (25 m x 50 m) of 0.74 ± 0.09(SE), 95% CI 0.56–0.91; Fig. 2.5). The systematic survey estimate of 30,800 burrows (95% CI 27,700–32,400; Appendix 2) lies within the 95% CI (20,700–71,100) from the random (GVE) transect estimate.

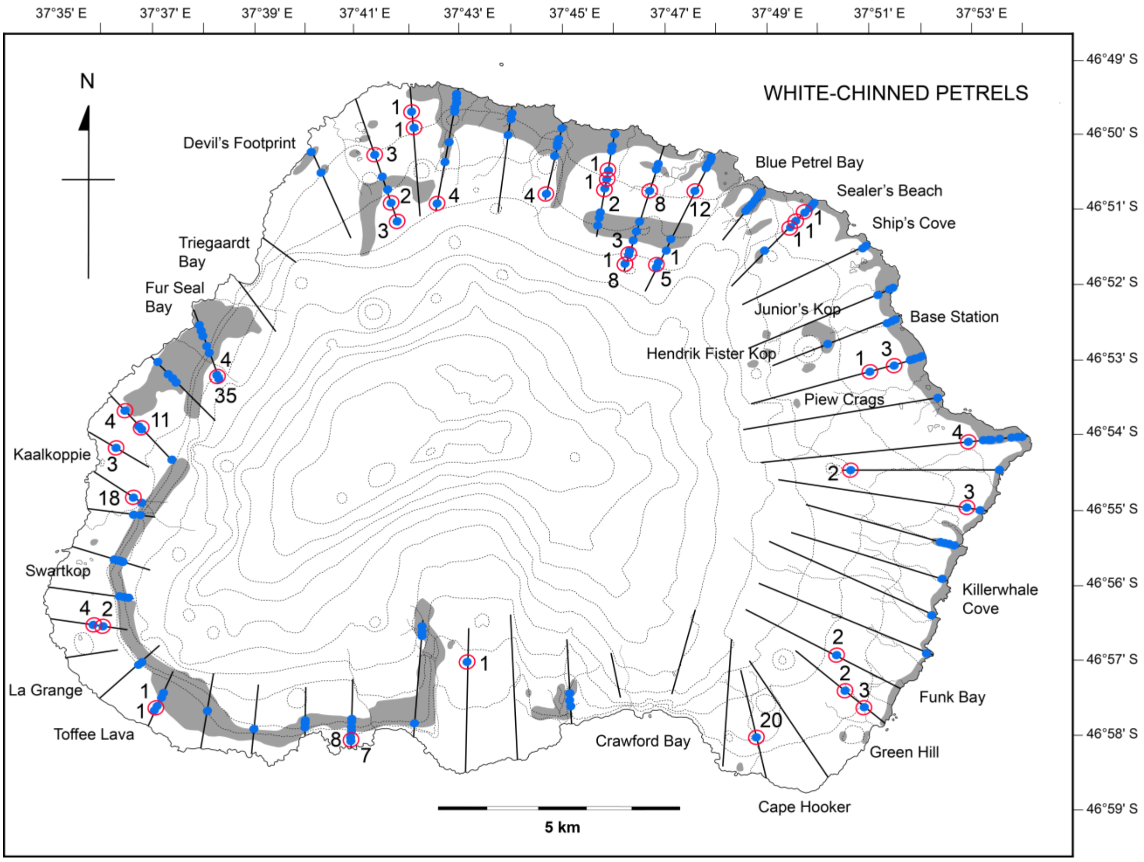


Figure 2.6. The distribution of white-chinned petrel burrows at Marion Island (adapted from Appendix 2), assessed using two sampling techniques: systematic in 2009 (whole island counts, grey shading) and random in 2015 (52 strip transects, total 144 km, black lines running inland from the coast). The blue dots indicate the locations of burrows within the 25 m wide strip transects; those circled in red were not recorded in the systematic survey (numbers indicate burrow counts at these sites).

Blue petrels were recorded at 29 sites along 13 of the 52 transects (25%) containing a total of 2,680 burrows (92 ± 108 burrows per site, range 5–500, Table 2.1). Five small sites missed in the systematic survey in 2012 were detected (5–50 burrows per site, total 120 burrows, 4% of total burrow count, Fig. 2.7). Estimates from 32 model combinations (<200 m) ranged from 102,500 burrows (VEA) to 152,800 burrows (GVES, Table 2.1, Appendix 2.5), 44% (31–83%) less than the systematic estimate of 214,700 burrows (Appendix 1) (Table 2.1). None of the 32 models appeared to accurately represent the habitat preferences of blue petrels, but based on field experience from the systematic survey I selected the combination of habitat attributes *geology-vegetation-elevation-aspect* (GVEA, Appendix 2.5) as the closest fit. This combination showed most burrows (>95%) occurred below 100 m elevation, 25% occurred in old weathered black lavas and 19% on Eastern Succession Fellfield and Mire-Slope. Using this combination I estimated there were 124,100 blue petrel burrows (95% CI) on Marion Island in 2015 (95% CI 40,500–191,000; CI based on mean blue petrel burrow counts per plot (25 m x 50 m) of 2.49 ± 0.79 (SE), 95% CI 0.81–3.83; Fig. 2.7). The systematic survey estimate of 214,700 burrows (95% CI 168,300–261,300; Appendix 1) lies outside the 95% CI (40,500–191,000) from the GVEA transect best estimate.

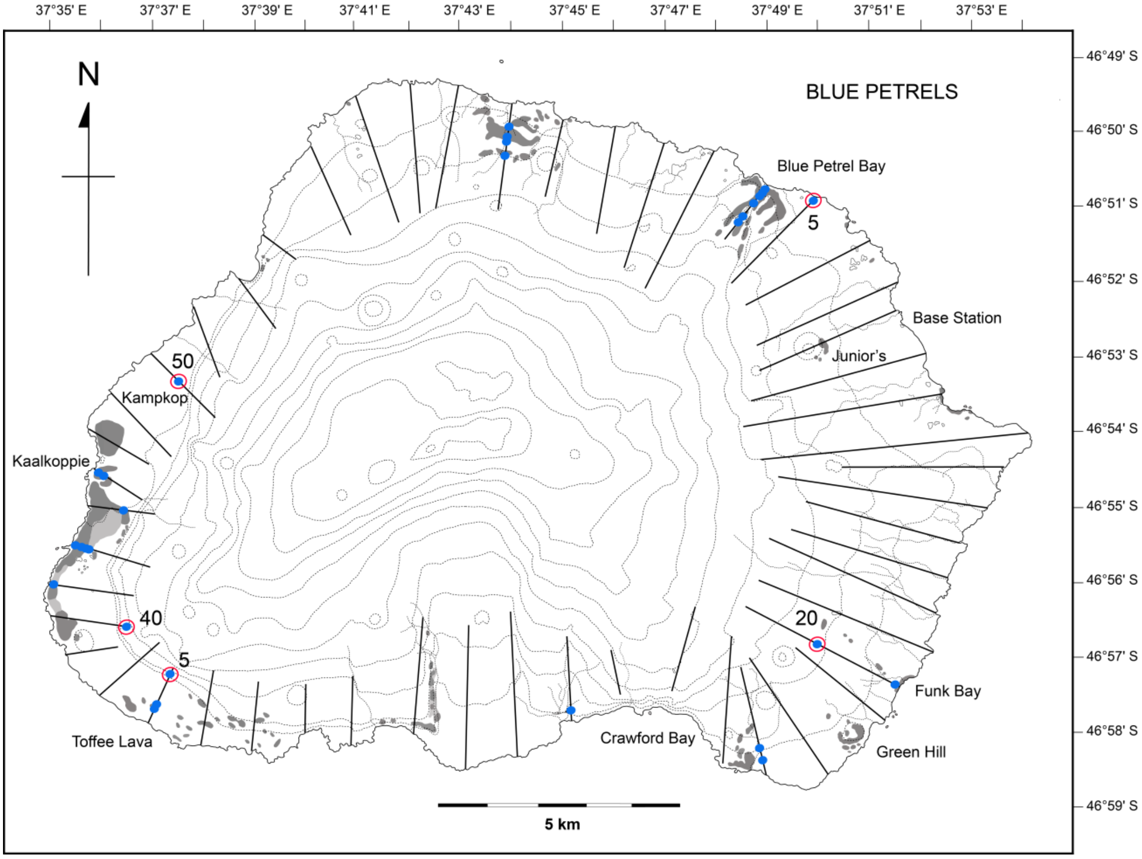


Figure 2.7. The distribution of blue petrel burrows at Marion Island (adapted from Appendix 1), assessed using two sampling techniques: systematic in 2012 (whole island counts, light shade = low and medium density, dark shade = high and very high density) and random in 2015 (52 strip transects, total 144 km, black lines running inland from the coast). The blue dots indicate the locations of burrows within the 25 m wide strip transects; those circled in red were not recorded in the systematic survey (numbers indicate burrow counts at these sites).

Discussion

Great-winged petrel distribution and population estimate on Marion Island

Rand (1954) reported that in the early 1950s "many [great-winged petrel] nests were isolated but others were grouped on tussock slopes and ridges (e.g. on Long Ridge) where the soft soil was easily worked" and that "burrows were usually sited where surface water could not settle" (p 194). van Zinderen Bakker Jr (1971) later noted "it is thought that the distribution on the islands is greatly influenced by the occurrence of the black lava humps in which nests were found" (p 167). Schramm (1986) studied the nest site preferences of six burrow-nesting petrel species on Marion and found great-winged petrels preferred nesting in deep, dry soils such as the inland vegetated slopes and the vegetated lava hummocks. The first population estimate was based on the field experience of Ryan and Bester (2008), who estimated 10,000 pairs on Marion. Although the distribution of great-winged petrel burrows around the island was largely unknown, the area around the base station (~300 ha) had been extensively explored. In that area, great-winged petrel burrows generally occur singly with the occasional larger aggregation in areas of particularly favourable habitat (Chapter 5).

My survey confirmed that great-winged petrel burrows are widely distributed at low densities around Marion Island, with more than half of the 237 sites where they were found having only one or two burrows. Habitat and elevation proved to be the best predictors of burrow distribution with birds favouring the deep soils (as found by Schramm 1986) associated with the oldest black lavas in the Fellfield and Mire-slope habitats (44% of burrows), especially between 50–150 m elevation (51% of burrows). Even with these predictors, the patchy and dispersed distribution of burrows would make a systematic survey of 293 km^2 for all great-winged petrel burrows impractical and very time consuming on such a large island. My objective was to establish population estimates prior to a mouse eradication as a baseline for future monitoring. Random sampling proved to be a practical, time-effective approach to estimate the numbers of this widely dispersed petrel, but its usefulness is limited by the large variances resulting from truly randomised (rather than targeted) sampling. Large variances restrict ones ability to detect small changes in future surveys, but a large increase or decrease (i.e. non-overlapping confidence intervals) in the number of burrows would likely be evident.

Survey design and effort

The design of burrow-nesting petrel surveys on remote oceanic islands requires careful consideration of multiple factors which will likely influence the accuracy and precision of the results. Survey effort may be constricted by resources (e.g. time on islands, budgets and availability of personnel) and the island structure (e.g. accessibility, vegetation height and density; see Schumann *et al*. 2013). On small islands (<200 ha) where burrow distributions are fairly uniform within clearly defined habitats (e.g. 143 ha Moutohora in New Zealand, Imber *et al*. 2003a) a systematic survey is practical to achieve accurate and precise estimates with a modest survey effort. However, larger islands where burrow-nesting petrels have patchy distributions and specific habitat associations (e.g. Marion Island (293 km^2), Schramm 1986; Campbell Island (112 km^2), Parker *et al*. 2017) usually require a substantial survey effort.

The survey effort for this random transect survey was substantially less (62 person-days, Table 2.2) than the systematic surveys of white-chinned (83, Appendix 2) and blue petrels (121, Appendix 1). However, the systematic surveys have narrower confidence intervals than the random survey results. Although systematic surveys are more laborious, involving extensive coverage of the island to search for colonies and detailed mapping of each colony found, systematic surveys ultimately provided better comprehension of the variance in densities and occupancies in different habitats than would have been evident through a random transect survey. However, I also show that systematic surveys are not always possible, particularly for species that are widely dispersed at low densities on large islands, like the great-winged petrel.

Table 2.2. Summary of the survey effort to estimate the population sizes of great-winged petrels in 2015 (52 random transects 25 m wide, total 144 km) compared to systematic whole island counts of white-chinned and blue petrels at Marion Island in 2009 (Appendix 2) and 2012 (Appendix 1).

Survey effort	**Great-winged petrel**	**White-chinned petrel**	**Blue petrel**
Survey type	random transect	systematic	systematic
Main survey period (days)	14 April - 1 May (17)	16 April - 5 May (19)	18 April - 6 May (18)
Main survey team	2–4 people (1–2 pairs)	3 people	5 people
Days to complete the main survey	15	17	15
Main survey person-days	48	50	72
Finish-off survey days (1 person)	0	12	18
Occupancy trials (n)	14 days (13)	21 days (18)	31 days (33)
Total person-days	62	83	121

Sources of error and the ability to detect future trends

Establishing good baseline estimates of burrow-nesting petrel populations requires as much ‘real’ data as possible (i.e. ideally, 100% coverage of the habitat a species uses). In reality this is not always possible or practical on large or remote uninhabited islands, but identifying and limiting the sources of error when planning a survey can improve accuracy and reduce the error margins (Parker and Rexer-Huber 2015), ultimately improving the ability to detect trends in population sizes over time. On Marion Island the relatively flat terrain allows for easy access to 95% of the island and the very low vegetation allows for fairly reliable burrow detection within the 25 m strip transects. This approach would be ineffective in the many cases where deep vegetation inhibits detection of burrow entrances, or in more mountainous terrain. Solutions to ensure high burrow detection rates in spite of terrain and/or deep vegetation can include shorter, narrow transects (2 m wide) or small plots (e.g. Rexer-Huber *et al*. 2017; Lawton *et al*. 2006; Rayner *et al*. 2007a, b; Parker *et al*. 2017).

The availability and use of accurate GIS habitat maps will enable large swathes of unsuitable habitat attributes to be excluded from future surveys (e.g. grey lava, recent black lava, scoria cones) on Marion Island. Large petrel burrows also peter out around 350 m elevation (while the island extends to 1,240 m) since the higher elevation areas (>600 m) are unsuitable for most large burrow-nesting petrel species, which further focussed the extrapolation area. The wide variance in the overall burrow estimate for great-winged petrels (32,400, 95% CI 15,800–46,300) is strongly influenced by the skewed distribution of burrows within the sample transects. The proportion of nests with signs of recent activity varied widely (average 48 ± 29%, range 10–94%), however reasons for this variability was not evident. In future great-winged petrel surveys, this could perhaps be addressed by additional checks and increased sampling at all occupancy trial sites.

Although extrapolating burrow densities by habitat attributes in multiple combinations did produce island estimates, assessing the accuracy of these estimates required analysing the burrow counts in standardised plots. I show how the standard error around the mean burrow count per plot is largest for blue petrels (clustered) and smaller for the more dispersed species (Fig. 2.7).

White-chinned petrels

Overall the random transects intersected most of the large areas where concentrations of white-chinned petrels were found in 2009 (Appendix 2). However, some areas were partly or entirely missed using random transects (e.g. *Blechnum* slopes below Junior's Kop, Hendrik Fister Kop and Piew Crags). Although burrow distribution was broadly represented by the random transects, the total white-chinned petrel burrow estimates based on model extrapolations were all higher than the systematic estimate, by 31–83%. The greater population estimate in 2015 might reflect in part the location of burrows outside areas identified in 2009 (18% of all burrows) as well as ongoing growth in this species' population on Marion Island (Chapter 3). However, it is also likely that the patchy distribution can skew results when random sampling is extrapolated across all suitable habitat from a relatively small sampling base (2.39% of area <450 m elevation).

Blue petrels

Random transect sampling greatly underestimated the number of blue petrel burrows around the island, giving 102,500–152,800 burrows compared to the 214,700 estimated from systematic counts (Appendix 1). This resulted from some key areas of the blue petrel distribution falling between transects and being omitted entirely (e.g. Green Hill and Grey-headed Ridge). Other key areas were intersected by transects, but the more densely-clustered colonies were missed (e.g. south of Kampkoppie, La Grange, Junior's Kop). Overall, the results were skewed by the small number of sites encountered by random transects, which failed to represent the habitat attributes favoured by blue petrels, as well as the range of sites where blue petrels are known to breed. These results highlight the usefulness of targeted sampling for species with highly clustered distributions.

Global population estimate of great-winged petrels

Great-winged petrels breed at six island groups in the Southern Oceans (Table 2.3), but the global population is poorly known since most breeding sites have not been accurately surveyed. On Gough Island, Swales (1965) described great-winged petrels as “abundant” (p 29) and as being seen “in large numbers in April 1956” (p 215) around The Glen on the east coast of the island and at higher altitudes. However field workers have been unable to locate any great-winged petrel nests over the last 30 years on Gough Island and it is very likely the population has been severely affected by mouse predation, and may even be extinct there (Chapter 4). Although formerly common on Tristan da Cunha, the impacts of harvesting chicks (Elliot 1953; Richardson 1984) and likely predation of chicks and eggs by introduced rats have greatly reduced the population, and occupied burrows are now uncommon (pers. obs. 2015–2018). On Marion Island, feral cats depredated chicks as a winter food source (van Aarde 1980) before cat eradication in 1990, but numbers have not increased as expected and mouse predation is the most likely explanation for the limited recovery over the last two decades (Chapter 5). Cats and rats extirpated great-winged petrels from Île aux Cochons and Île de la Possession in the Crozet Archipelago (Jouventin *et al*. 1984). Currently, most large populations of great-winged petrels occur at islands lacking introduced rodents: Predator-free islands in Golfe du Morbihan (Kerguelen), Île de l'Est (Crozet), Prince Edward Island and Eclipse Island (off Australia’s south-west coast).

Conclusions

Random sampling protocols remain a standard tool for estimating burrowing petrel populations. However, where resources and time allow, systematic surveys are more reliable methods to estimate populations of species with highly clustered distributions, even if only as a one-off to obtain a ‘big picture’ understanding of the population on an island. Repeating such intense systematic surveys for long-term monitoring of changes may not be feasible, but the information gained from a systematic survey could be used to design a less-intensive random stratified approach for longer-term monitoring. The interval between surveys is also important to consider, because most seabirds have conservative life histories characterised by low reproductive rates and long lifespans, making it difficult to detect increases in population size over the short- to medium-term (although there are exceptions, e.g. the steady increase in spectacled petrels *Procellaria conspicillata* surveyed in 1999, 2004 and 2009 at Inaccessible Island; Ryan and Ronconi 2011). However, more frequent surveys may be appropriate for declining populations, as even quite small increases in mortality can lead to rapid population declines.

Table 2.3. Great-winged petrels are known to breed at six island groups in the Southern Oceans.

Location (area km^2)	**Breeding pairs**	**Count method**	**Year**	**Source**	**Introduced mammals**
Tristan da Cunha Archipelago (179)					
Tristan da Cunha (96)	3,000-4,000	Estimate based on field experience	2015-2018	a, b	black rats, mice, (cats eradicated 1970s)
Gough (65)	<100 or probably extinct	Estimate based on field experience	2014-2017	b	mice
Prince Edward Islands (334)					
Prince Edward Island (44)	20,000	Estimate based on field experience	2008	c	no introduced predators
Marion Island (293)	14,000	Transects & density extrapolation	2015	d	mice (cats eradicated 1990)
Crozet Archipelago (352)					
Île de l'Est (130)	10s of 1,000s	Estimate based on field experience	1984	e	rabbits
Île des Pingouins (3)	Breeding observed (no counts)	Estimate based on field experience	1984	e	no introduced predators
Île des Apôtres (2)	Breeding observed (no counts)	Estimate based on field experience	1984	e	no introduced predators
Île de la Possession (150)	Probably extinct	No recent surveys	-	e	black rats
Île aux Cochons (67)	Probably extinct	No recent surveys	-	e	cats, mice, rabbits
Kerguelen Archipelago (7,200)					
Islands in Golfe du Morbihan (23)	Unconfirmed numbers	Distance sampling	Recent	f	some have black rats, mice, rabbits
	(previous estimates ~100,000)		1980s	g	
Amsterdam and Saint Paul Islands (63)					
Amsterdam (55)	<100 or probably extinct	Estimate based on field experience	1990s	h	rats, mice, cats, pigs, wild cows
Saint Paul Island (8)	<50 pairs (increasing)	Estimate based on field experience	1999	i	mice, possibly still rabbits (rats eradicated 1999)
Roche Quille Stack (<0.5)	<50 pairs	Estimate based on field experience	1999	i	no introduced predators
Islands off Australia's S/SW coasts (<5)					
Albany District:					
Eclipse Island (0.6)	10,000-15,000	1980s island survey & recent observations	Recent	j	rabbits
Breaksea, Bald, Coffin, Gull Islands	3,000-4,000	1980s island survey & recent observations	Recent	j	some have rabbits, skinks
Archipelago of the Recherche:					
Goose,Hood, Wilson, Bellinger, Middle & (Termination) Islands	1,300	1980s island survey & recent observations	Recent	j	some have rabbits
Boxer, Christmas & Douglas Islands	Unconfirmed numbers	Recent observations	Recent	k	cats, rats, reptiles, crazy ants
Mistaken Island	Extinct between 1911 and 1921	Recent observations	Recent	k	formerly rabbits (eradicated 1980)
Global breeding population	**150,000 pairs**				

Data sources: (a) Flood & Zufelt *in press* 2018 (at-sea obs.); (b) BJD, PGR pers. obs.; (c) Ryan & Bester (2008); (d) this study; (e) Jouventin *et al*. (1984); (f) Pers. comm. Karine Delord & Adrien Chaigne 2018; (g) del Hoyo *et al*. 1992; (h) Micol & Jouventin (1995); (i) Micol & Jouventin (2002); (j) Johnstone & Storr, 1998; Pers. comm. Johnstone 2018; (k) Pers. comm. Johnstone 2018

Appendix 2.1. Descriptions of geological and vegetation classes on sub-Antarctic Marion Island used in models to predict the abundance and distribution of burrow-nesting petrels in 2015 (adapted from Boelhouwers *et al*. 2008)

Geological classes	
Black lava types 1-4	extensive areas of rocky broken black lava flows, classed by age:
Type 1	- oldest flows which are now vegetated lava hummocks
Types 2 & 3	- partly vegetated lava hummocks
Type 4	- youngest, raw black lava
Recent lava flows (post-1980)	e.g. at Kaalkop on west coast
Eastern and Western successions	grey lava successions radiating from the centre of the island
Holocene scoria cones	resulting from several volcanic eruptions subsequent to glaciations
Wind-blown volcanic ash	ash from volcanic eruptions subsequent to glaciations
Vegetation classes	
Cinder Cones	largely unvegetated, occasional patches of *Acaena magellanica* creeper and isolated cushion plants *Azorella selago*
Coastal Vegetation	coastal slopes and flat areas with two main habitats: - **salt spray areas** - dominated by extensive low herbfields of *Cotula plumosa*, *Crassula moschata* & sprawling *A. selago* **- biotic areas** - fertilised by seal and seabird colonies dominated by tussock grass *Poa cookii* & sedge *Uncinia compacta*
Fellfield	- loose broken black lava dominated by mosses *Sphagnum* species, cushion plants *A. selago* & *Agrostis* grasses - steep slopes at lower altitudes dominated by *Blechnum penna-marina* ferns and *A. magellanica* creepers
Mire-Slope Vegetation	- mires: wet, relatively flat boggy areas on black lava coastal plains & on grey lava ridges with *Sphagnum* mosses & *Agrostis* grasses - steep vegetated slopes: well-drained soils on grey lava ridges dominated by creeping stems of *B. penna-marina* ferns & *A. magellanica* creeper which form large soft mats of vegetation
Polar Desert	at elevations >650 m, shallow or no soil cover, lichens & mosses

Appendix 2.2. The estimated number of **great-winged, white chinned and blue petrel** burrows on Marion Island in 2015 using burrow densities estimated from 52 random transects (25 m wide, total 144 km). Burrow densities and five associated habitat attributes (G, V, S, A, E) were grouped across multiple combinations in 32 models to generate island estimates.

Model	**Great-winged petrel**	**White chinned petrel**	**Blue petrel**
Null	35,336	56,391	116,305
Geology (G)	33,710	50,034	121,156
Vegetation (V)	33,192	50,660	111,962
Slope (S)	35,231	53,189	113,384
Aspect (A)	34,449	51,794	112,638
Elevation (E)	33,557	48,612	116,104
GV	32,417	49,862	119,297
GE	33,659	46,794	130,326
GA	35,012	52,189	122,525
GS	36,545	52,642	131,250
VE	35,559	52,264	112,523
VA	33,287	50,015	111,276
VS	34,587	51,555	117,362
EA	33,293	47,209	111,130
ES	34,579	48,961	113,787
AS	36,836	55,417	111,122
GVE	**32,386**	**48,517**	129,844
GVA	33,528	49,314	122,724
GVS	32,559	47,473	127,281
GEA	33,596	44,806	112,747
GES	33,502	46,718	138,968
GAS	34,983	48,715	120,405
VEA	33,348	47,111	102,510
VES	36,047	52,335	143,230
VAS	35,473	52,444	107,484
EAS	32,318	46,273	112,196
GVEA	33,472	47,482	**124,101**
GVES	29,910	44,825	152,825
GVAS	32,198	45,154	121,860
GEAS	28,721	41,017	115,955
VEAS	33,254	49,063	106,170
GVEAS	28,515	40,237	113,952
Average	33,596	49,033	119,512
SD	1,936	3,637	10,977
Minimum	28,515	40,237	102,510
Maximum	36,836	56,391	152,825
Median	33,543	49,012	116,205

Appendix 2.3. The estimated number of **great-winged petrel** burrows on Marion Island in 2015. Burrow densities were estimated from 52 random transects (25 m wide, total 144 km) and extrapolated to habitat attributes: Geology-Vegetation-Elevation (GVE)

Sub-Antarctic habitat	**0-50 m**	**50-100 m**	**100-150 m**	**150-200 m**	**200-250 m**	**250-300 m**	**300-350 m**	**Total**	
Black Lava 1 Coastal Vegetation	341	0	0	0	0	0	0	341	1%
Black Lava 1 Fellfield	0	1,424	0	360	482	1,168	633	4,066	13%
Black Lava 1 Mire-Slope Vegetation	3,394	3,304	2,257	1,346	27	0	0	10,327	32%
Black Lava 2 Coastal Vegetation	162	0	0	0	0	0	0	162	<1%
Black Lava 2 Fellfield	0	41	163	33	0	583	0	819	3%
Black Lava 2 Mire-Slope Vegetation	235	1,502	716	402	0	0	0	2,854	9%
Black Lava 3 Fellfield	0	257	53	187	1,121	78	0	1,695	5%
Black Lava 3 Mire-Slope Vegetation	55	462	1,881	1,030	108	0	0	3,537	11%
Black Lava 4 Coastal Vegetation	39	146	0	0	0	0	0	185	<1%
Black Lava 4 Fellfield	0	0	341	183	0	0	0	524	2%
Black Lava 4 Mire-Slope Vegetation	197	1,160	447	190	48	0	0	2,042	6%
Eastern Succession Coastal Vegetation	62	63	0	0	0	0	0	125	<1%
Eastern Succession Fellfield	583	787	537	363	790	1,206	0	4,266	13%
Eastern Succession Mire-Slope Vegetation	0	318	345	501	0	0	0	1,163	4%
Eastern Succession Cinder Cones	0	0	257	0	0	0	0	257	<1%
Western Succession Coastal Vegetation	19	0	0	0	0	0	0	19	<1%
Total burrow counts	5,087	9,463	6,997	4,595	2,576	3,035	633	**32,386**	
16%	29%	22%	14%	8%	9%	2%		100%	

Appendix 2.4. The estimated number of **white-chinned petrel** burrows on Marion Island in 2015. Burrow densities were estimated from 52 random transects (25 m wide, total 144 km) and extrapolated to habitat attributes: Geology-Vegetation-Elevation (GVE)

Sub-Antarctic habitat	**0-50 m**	**50-100 m**	**100-150 m**	**150-200 m**	**200-250 m**	**250-300 m**	**300-350 m**	**350-400 m**	**400-450 m**	**Total**	
Black Lava 1 Coastal Vegetation	7,100	0	0	0	0	0	0	0	0	7,100	15%
Black Lava 1 Fellfield	0	2,305	0	818	602	844	742	362	0	5,672	12%
Black Lava 1 Mire-Slope Vegetation	7,118	6,235	135	601	0	0	0	0	0	14,089	29%
Black Lava 2 Coastal Vegetation	108	0	0	0	0	0	0	0	0	108	<1%
Black Lava 2 Fellfield	0	284	1,592	0	0	466	0	0	0	2,342	5%
Black Lava 2 Mire-Slope Vegetation	47	1,612	28	186	0	0	0	0	0	1,872	4%
Black Lava 3 Coastal Vegetation	267	0	0	0	0	0	0	0	0	267	<1%
Black Lava 3 Fellfield	0	449	0	0	133	0	0	0	0	582	1%
Black Lava 3 Mire-Slope Vegetation	887	1,284	493	98	0	0	0	0	0	2,762	6%
Black Lava 4 Fellfield	0	0	1,213	61	0	0	0	0	0	1,274	3%
Black Lava 4 Mire-Slope Vegetation	1,974	548	298	54	0	0	0	0	0	2,874	6%
Eastern Succession Cinder Cones	0	0	298	0	0	0	0	0	0	298	<1%
Eastern Succession Coastal Vegetation	3,899	38	0	0	0	0	0	0	0	3,936	8%
Eastern Succession Fellfield	2,148	569	0	0	136	0	324	0	0	3,177	7%
Eastern Succession Mire-Slope Vegetation	330	423	172	143	0	0	0	0	0	1,069	2%
Western Succession Fellfield	0	215	0	0	0	879	0	0	0	1,095	2%
Total burrow counts	23,877	13,962	4,229	1,961	871	2,189	1,066	362	0	**48,517**	
	49%	29%	9%	4%	2%	5%	2%	1%	0%		100%

Appendix 2.5. The estimated number of **blue petrel** burrows on Marion Island in 2015. Burrow densities were estimated from 52 random transects (25 m wide, total 144 km) and extrapolated to habitat attributes: Geology-Vegetation-Elevation-Aspect (GVEA)

Sub-Antarctic habitat	**0-50 m**	**50-100 m**	**100-150 m**	**150-200 m**	**Total**	
Black Lava 1 Mire-Slope Vegetation E	8,673	0	0	0	8,673	7%
Black Lava 1 Mire-Slope Vegetation N	0	2,781	0	0	2,781	2%
Black Lava 1 Mire-Slope Vegetation NE	190	22,048	0	0	22,238	18%
Black Lava 1 Mire-Slope Vegetation NW	850	0	0	0	850	1%
Black Lava 1 Mire-Slope Vegetation SW	3,047	0	0	0	3,047	2%
Black Lava 2 Mire-Slope Vegetation NW	0	557	0	0	557	0%
Black Lava 2 Mire-Slope Vegetation SW	0	5,581	0	0	5,581	4%
Black Lava 3 Coastal Vegetation N	620	0	0	0	620	0%
Black Lava 3 Coastal Vegetation W	0	190	0	0	190	0%
Black Lava 3 Mire-Slope Vegetation N	40,572	3,080	0	0	43,652	35%
Black Lava 3 Mire-Slope Vegetation NE	0	0	0	2,730	2,730	2%
Eastern Succession Coastal Vegetation NE	3,426	0	0	0	3,426	3%
Eastern Succession Fellfield E	0	0	1,532	0	1,532	1%
Eastern Succession Fellfield N	1,339	0	0	0	1,339	1%
Eastern Succession Fellfield NE	0	0	1,642	0	1,642	1%
Eastern Succession Fellfield NW	1,327	2,946	0	0	4,273	3%
Eastern Succession Fellfield S	0	7,080	0	0	7,080	6%
Eastern Succession Mire-Slope Vegetation N	0	2,605	0	0	2,605	2%
Eastern Succession Mire-Slope Vegetation W	0	11,285	0	0	11,285	9%
Total burrow counts	60,044	58,153	3,173	2,730	**124,101**	
	48%	47%	3%	2%		100%

Chapter 3

Modest increases in densities of burrow-nesting petrels following the removal of cats *Felis catus* from sub-Antarctic Marion Island

A white-chinned petrel *Procellaria aequinoctialis* sitting outside its burrow at dusk on Marion's west coast (photo Ben Dilley).

This chapter is broadly based on this publication:

Dilley, B.J., Schramm, M. and Ryan, P.G., 2017. Modest increases in densities of burrow-nesting petrels following the removal of cats (Felis catus) from Marion Island. *Polar Biology, 40*(3), pp.625-637.

Author contributions: BJD, MS & PGR planned the field work at the study site; BJD completed the survey, analysed the data and wrote the draft; PGR assisted with data analyses, data presentation and manuscript edits/preparation; MS contributed to drafts.

Chapter 3: Modest increases in densities of burrow-nesting petrels following the removal of cats *Felis catus* from sub-Antarctic Marion Island

Abstract

Introduced predators are one of the main threats facing seabirds breeding on oceanic islands. Cats *Felis catus* were introduced to sub-Antarctic Marion Island (293 km^2) in 1949, and by the 1970s some 2,000 cats were killing about 450,000 seabirds per year, greatly reducing burrowing petrel populations. Cats were eradicated by 1991, but house mice *Mus musculus*, remain. The densities of utilised petrel burrows (eight species) were estimated in 2013 by systematically searching for their burrows in 741 10x10 m sample quadrats in the north-eastern sector of Marion Island, repeating the sampling design and methods used by Schramm in 1979. The mean burrow densities and 95% CIs were compared between surveys by species for the different habitat and vegetation types, with non-overlapping CIs considered indicative of a change in burrow density. The combination of cats being eradicated and the potential for immigration from nearby Prince Edward Island (free of introduced mammals) could promote a multi-fold increase in petrel numbers over the last two decades, however burrow densities at Marion have increased by only 56% since 1979. White-chinned petrels *Procellaria aequinoctialis* showed the greatest increase, despite being listed as Vulnerable due to incidental mortality on fishing gear at sea. The recovery of other summer-breeding species decreased with decreasing body size, and winter-breeding species showed even smaller recoveries, similar to patterns of breeding success at Gough Island, where mice are major predators of petrel chicks and eggs. There is compelling evidence that mice predation impacts are influencing the recovery of Marion's petrel populations.

Introduction

Many seabird species are threatened with extinction, and one of the major threats, particularly for oceanic species, is the introduction of mammalian predators onto their breeding islands (Croxall *et al*. 2012). The sub-Antarctic Prince Edward Islands (46°54'S, 37°45'E) in the southwest Indian Ocean provide a sobering example of the consequences of such introductions, which extend beyond the impacts on seabird populations to affect the structure and functioning of the islands' terrestrial ecosystems (Chown and Smith 1993; Smith *et al*. 2002). The Prince Edward Island group comprises two islands: Marion (293 km^2) and Prince Edward (44 km^2). They support 29 species of breeding seabirds (Ryan and Bester 2008), but Marion Island has a much larger complement of introduced species, following the establishment of a weather station on the island in 1948 (Chown and Froneman 2008). House mice *Mus musculus* were introduced accidentally to Marion Island, most likely by sealers or shipwrecks in the early 19th century (Watkins and Cooper 1986). Domestic cats *Felis catus* were taken to the island's weather station in 1949 to control mice (van Aarde 1977), but they soon turned feral and started killing the island's seabirds (Rand 1954). By the mid-1970s an estimated 2,000 cats were killing some 450,000 birds per year, most of which were burrow-nesting petrels (van Aarde 1980). Population densities were reduced more than 20-fold (Schramm 1986), and some petrels were apparently extirpated (van Aarde 1980; Ryan and Bester 2008). By comparison, Prince Edward Island has not had any introduced mammals (Gleeson and van Rensburg 1982).

The population sizes of burrow-nesting petrels (Procellariidae, Pelecanoididae and Hydrobatidae) at the Prince Edward Islands are poorly known relative to surface-nesting species (Diomedeidae and Spheniscidae; Crawford *et al*. 2009; Ryan *et al*. 2009a). At least nine species of burrowing petrels breed on Marion Island, but two small species probably were extirpated (black-bellied storm petrel *Fregetta tropica* and common diving petrel *Pelecanoides urinatrix*) and populations of other species were greatly depressed by cat predation (Ryan and Bester 2008). A multi-faceted cat eradication programme was started in the late 1970s, and by 1991 the last cat had been killed (Bester *et al*. 2002), allowing burrow-nesting petrel numbers to recover. There have only been two detailed estimates of burrow-nesting petrel population sizes at the Prince Edward Islands based on burrow densities (Schramm 1986; Appendix 2). There has been little effort to assess how petrel populations at Marion Island responded following the eradication of cats. Initial indications were positive; following the removal of cats there were marked increases in the breeding success of burrowing petrels, especially the great-winged petrel *Pterodroma macroptera* which breeds in winter when cat predation pressure was most severe (Cooper and Fourie 1991; Cooper *et al*. 1995). However, recent evidence from sub-Antarctic skuas *Catharacta antarctica lönnbergi*, which are major predators of burrowing petrels, were less encouraging (Cerfonteyn and Ryan 2016). There is a significant relationship between food availability and reproductive success in skuas (Phillips *et al*. 1996), but numbers of skuas have decreased steadily at Marion Island since cats were eradicated, whereas their numbers at Prince Edward Island have remained stable (Ryan *et al*. 2009b).

At other islands, petrel populations have recovered rapidly following the eradication of terrestrial predators. For example, numbers of spectacled petrel *Procellaria conspicillata*, a species closely related to white-chinned petrels, have increased at roughly 7% per year following the disappearance of pigs from Inaccessible Island (Ryan *et al*. 2006; Ryan and Ronconi 2011). If similar recoveries have occurred at Marion Island since cats were eradicated, it could be expected that burrowing petrel populations would have increased 3–5–fold by 2013 (based on 5–7% per annum growth), and potentially even more given immigration from nearby Prince Edward Island (22 km to the NE), creating easily detectable signals of population recovery (Ryan *et al*. 2006). Such increases are not unrealistic, because densities of burrowing petrel nests on Marion Island at the height of the cat era were approximately 25 times lower (200 ha^{-1}) than those on Prince Edward Island (5,000 ha^{-1}; Schramm 1986). Schramm (1986) estimated the density of burrowing petrel nests in the northeast sector of Marion Island in 1979. I repeated Schramm's survey in 2013, more than 20 years after cats were removed from the island, to assess the extent to which petrel populations at Marion Island have recovered since cats were eradicated.

Study area and methods

Repeat survey

To estimate petrel breeding densities, I systematically searched for their burrows in 741 10x10 m sample quadrats in the north-eastern sector of Marion Island, repeating the sampling design and methods used by Schramm (1986) over the austral summer of 1979/80. Burrows were sampled at 13 sites from altitudes of 12 m to 373 m, covering an area of approximately 1,040 ha (Fig. 3.1) with five habitat types: steep vegetated slopes, coastal lowland, vegetated lava hummocks, partly vegetated lava hummocks and cinder slopes. At 11 of these sites, quadrats were arranged in blocks of three 10x10 m (*n* = 215) and laid out at 25 m intervals perpendicular to transect lines 200 m apart. At the remaining two sites, random transect lines (1,200 m) were chosen and 10x10 m quadrats (*n* = 96) sampled at 50 m intervals for more extensive coverage in the vegetated and partly-vegetated lava hummocks. The location of Schramm's (1986) sample quadrats were only crudely mapped in 1979. To accurately repeat the 1979 sampling protocol, Mike Schramm returned to Marion Island in April-May 2012 to identify the location of the original transects (Fig. 3.1). Sites were matched to the same habitats sampled in 1979 and marked with poles for future monitoring.

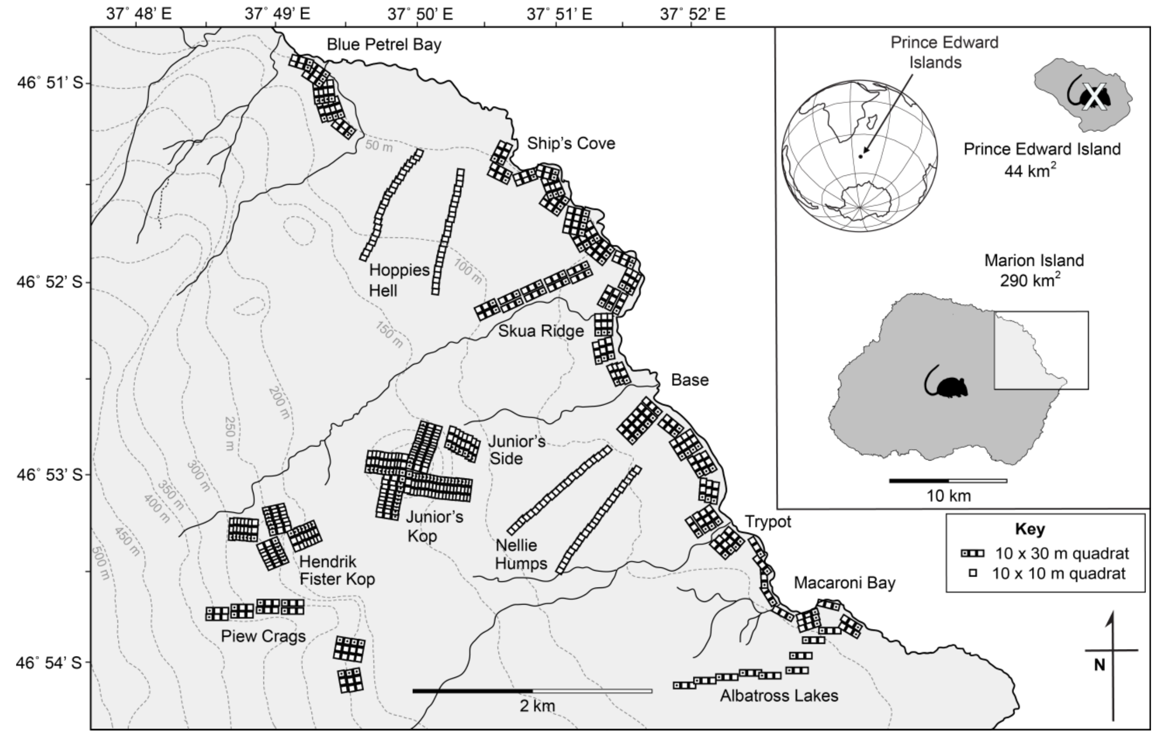


Figure 3.1. Study area in the northeast corner of Marion Island, showing the locations and arrangement of the 741 (10x10 m) sample plots. The insert shows the location of the Prince Edward Islands, with Prince Edward Island 22 km to the northeast of Marion.

The actual surveys to estimate burrow densities for all species were repeated in summer (29 January to 22 March 2013), when most burrowing petrels breed. At the time of the surveys, Salvin's prion *Pachyptila salvini* and blue *Halobaena caerulea*, soft-plumaged *Pterodroma mollis,* white-chinned

Procellaria aequinoctialis, South Georgian diving *Pelecanoides georgicus* and common diving petrels were still breeding, and Kerguelen petrel *Aphrodroma brevirostris* chicks had recently fledged (Fig. 3.2). During the repeat survey, the two winter breeding species (great-winged and grey petrels *Procellaria cinerea*) were not active as the timing of this repeat survey fell outside their breeding cycle. Therefore I intensively surveyed a ~300 ha area around Base for utilised nests of these two species during the winter of 2009 and 2012. A proportion of these burrows were monitored to record the birds breeding phenology for additional research projects. These studies provided a reasonable baseline for identifying recently active burrows of these two winter breeding species during the actual survey.

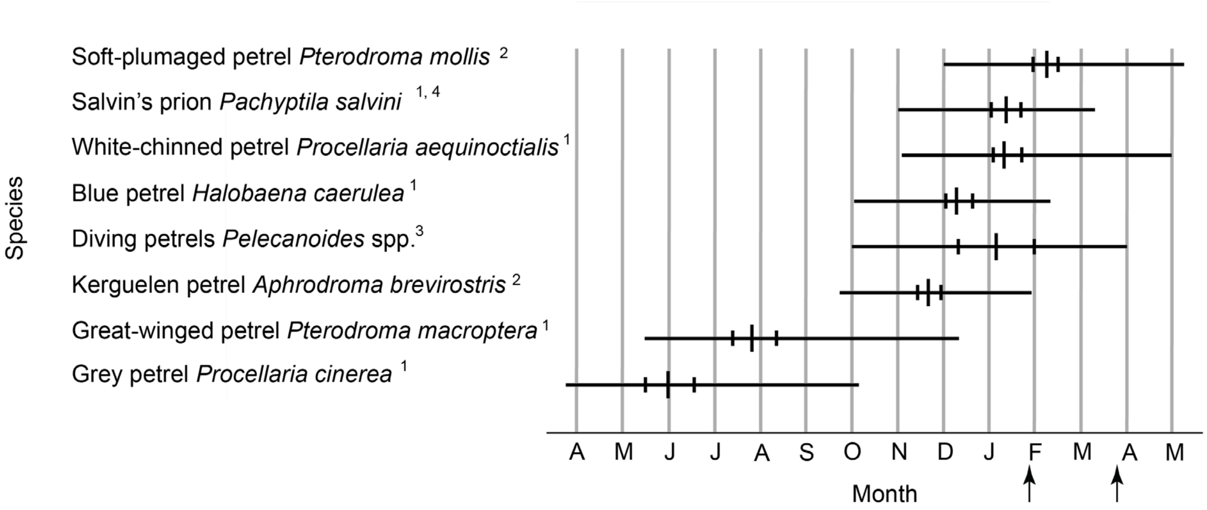


Figure 3.2. Breeding months of burrowing petrels at Marion Island. Vertical bars indicate the hatching periods and the central vertical bars give the average hatching dates. Arrows indicate the timing of the 2013 repeat survey of 741 (10x10 m) quadrats. Data sources: [1] Ben Dilley, FitzPatrick Inst. unpubl. data, Marion Island 2009–2014; [2] Schramm (1983); [3] assumed breeding period for diving petrels *Pelecanoides* spp., Payne and Prince (1979); [4] Berruti & Hunter 1985.

For each 10x10 m sample quadrat I recorded slope aspect (using a compass), soil depth (using a graduated 1-m metal rod), slope angle (using a clinometer), altitude (using a Garmin GPSmap62s), the dominant vegetation type (based on Huntley 1971, Smith 1976) and the habitat type based on the five habitat types described in detail by Schramm (1986) based on the classification by Verwoed (1971). In both surveys, these parameters were recorded once for each quadrat and assumed to be representative for all burrows within the quadrat (following methods described in Schramm (1986)).

Burrow identification

In both surveys, all the active burrows (with a bird present) and recently active burrows were identified and counted. Recently active burrows were identified by signs of fresh excavations, freshly cropped vegetation (used as nest lining), fragments of new egg shell (indicative of a failed nest), feathers, fresh guano, fresh footprints or disturbed moat water (clear, settled moat water signifies birds have not recently been through the entrance). Burrows which were overgrown or collapsed were not counted. During his return to Marion Island in April/May 2012, Mike Schramm showed me how he used the relative shape and size of the burrow entrance (Fig. 3.3) and the physical burrow characteristics (Appendix 3.1) to infer the species occupying a burrow. Further confirmation by probing the burrow with a stick to elicit a response (Appendix 2) or using play-back recordings or vocal impersonations was a reliable method for identifying white-chinned petrels (Berrow 2000; Appendix 2), blue petrels (Fugler *et al*. 1987; Crawford 1952), grey petrels (Barbraud *et al*. 2009; pers. obs. 2009, 2012, 2014) and Kerguelen petrels (pers. obs. 2012), whereas great-winged and soft-plumaged petrels tended to be less responsive. However, even among highly responsive species, some individuals are less likely to call back than others.

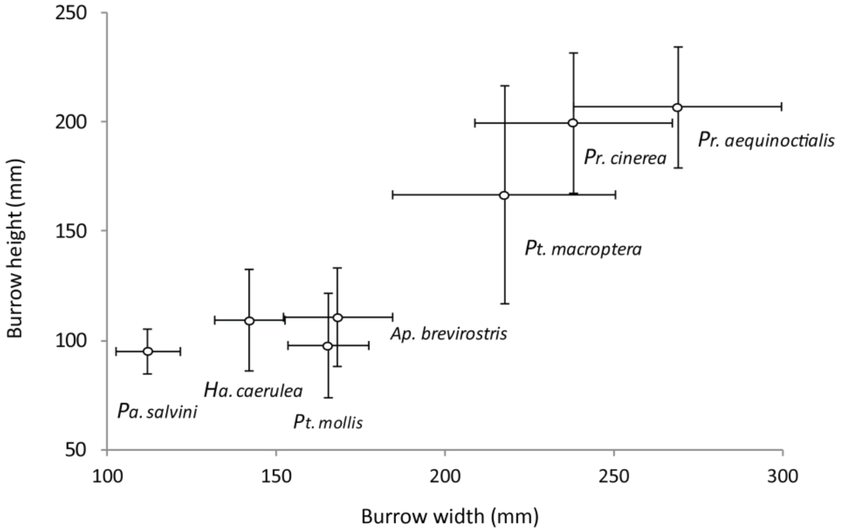


Figure 3.3. Average burrow entrance dimensions of Salvin's prion *Pachyptila salvini* (*n* = 17)[1], blue petrel *Halobaena caerulea* (*n* = 30)[1], soft-plumaged petrel *Pterodroma mollis* (*n* = 16)[2], Kerguelen petrel *Aphrodroma brevirostris* (*n* = 15)[2], great-winged petrel *Pterodroma macroptera* (*n* = 50)[1], white-chinned petrel *Procellaria aequinoctialis* (*n* = 50)[1] and grey petrel *Procellaria cinerea* (*n* = 51)[3]. Error bars represent ± 1 SD. Data sources: [1] This study; [2] Schramm (1983); [3] Ben Dilley, FitzPatrick Inst. unpubl. data, Gough 2014.

Data on vocalisations of petrels and on physical characteristics of their burrows (Appendix 3.1) were collected prior to the survey at study burrows in study colonies of grey, great-winged, white-chinned and blue petrels which I conducted at Marion in 2009/10 and in 2012/13. I did not collect further data on the physical characteristics during this repeat survey. For Kerguelen and soft-plumaged petrels I used data on burrow dimensions from Schramm (1983). Data on burrow dimensions of grey petrels were from my study colony on Gough Island in 2014, since grey petrels predominantly nest in caves on Marion, but also use burrows. In addition to these methods, I custom-built a burrow-scope with a high resolution conical pinhole camera, LED torch and an 18x21 cm colour monitor which provided a clear image of the inside of the burrow. Although the burrow-scope allowed for low-impact inspection of burrow contents, some burrows were too complex or deep to see the nest chamber from the burrow entrance. Burrows with two entrances that connected to one passage were counted as one and burrows on the edge of a quadrat were included if the burrow entrance was completely within the square. Using a combination of these burrow identification methods, I allocated a species to each utilised burrow in each quadrat. The resultant burrow densities in both surveys are a measure of how many utilised burrows were found, but since these were once off burrow checks, the results do not accurately quantify how many burrows contained breeding pairs or non-breeders, nor account for inter-annual variability in burrow occupancy.

Data analysis

Schramm (1986) reported burrow densities (burrows·ha^{-1}) of eight petrel species at 13 sites (total of 741 quadrats) as mean ± one standard deviation (SD) with the associated number of sample quadrats (0.01 ha) per habitat type and sample site (Appendix 3.2). From these data, I calculated the standard errors (SE = SD/$\sqrt{n}$) and 95% confidence intervals (CI = mean ± 2SE) from the mean burrow densities of each species for the five habitat types and seven vegetation types. Where 2013 data produced a negative CI, these data were bootstrapped (analysis was run using library *boot* (Canty and Ripley 2012) in R (R Core Team 2014) with 5,000 iterations). These mean burrow densities and 95% CIs were compared between surveys by species for the different habitat and vegetation types, with non-overlapping CIs considered indicative of an increase in burrow density. Means are presented ± SD unless otherwise indicated.

Mean body mass of each species (data from: Schramm 1983; Fugler *et al*. 1987; Payne and Prince 1979; Berruti and Hunter 1986; FitzPatrick Inst. unpubl. data) was log (ln) transformed to interpret the relationship between body mass and the apparent increase in burrow density between surveys, estimated as the density in 2013/density in 1979. Following methods described in Schramm (1986), the 2013 burrow density data were extrapolated to the larger study area (1,041 ha) by habitat type to estimate the number of burrows for each species. The percentage increase in the number of burrows between surveys was calculated as the difference in number of burrows between surveys/number of burrows in 1979.

Inter-specific differences in mean soil depth (mm) and slope angle (degrees) were tested using Kruskal-Wallis and post-hoc Tukey tests. I calculated the average direction which burrow entrances faced for each species using the package 'circular' (Agostinelli and Lund 2013) in R 3.1 (R Core Development Team 2014). I calculated 95% CIs from aspect data (bootstrapped with 1,000 iterations, using library *boot* (Canty and Ripley 2012) in R (R Core Team 2014)). Species with non-overlapping 95% CIs were considered to be significantly different and statistical tests were two-tailed with $p < 0.05$ as the cut-off for significance.

Results

Of the six petrel species recorded in both studies, only the summer breeding white-chinned petrels and Salvin's prions showed marked increases in densities from 1979 to 2013. Recovery of other summer-breeding species decreased with decreasing body size (Fig. 3.4), and winter-breeding species showed even smaller recoveries, with numbers of grey petrels apparently having decreased. Blue petrels, soft-plumaged petrels, Kerguelen petrels and great-winged petrels showed marginal increases in densities. Although overall burrow numbers in the study area (1,041 ha) increased by 56% between the two surveys, if I exclude white-chinned petrels the increase is 51%.

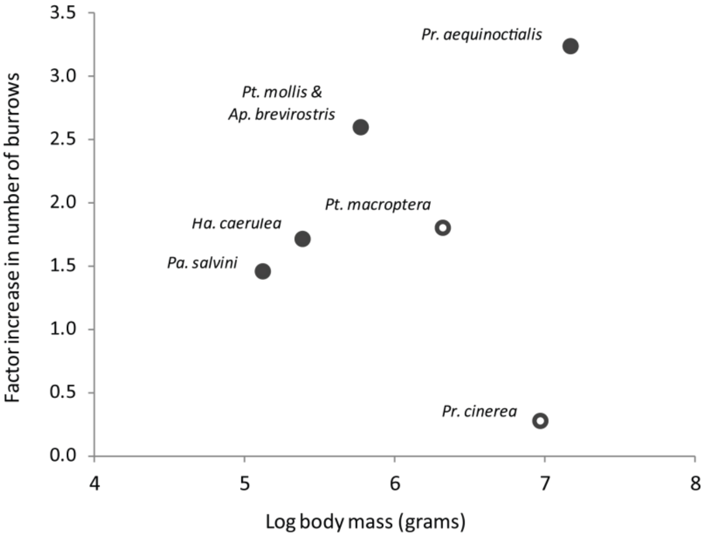


Figure 3.4. The relationship between body mass (g) and the increase in number of burrows in northeast Marion Island from 1979 to 2013. Data from Kerguelen *Aphrodroma brevirostris* and soft-plumaged *Pterodroma mollis* petrels are pooled due to the low number of burrows recorded and their similar burrow size. Winter breeders represented with open circles. Species names are Salvin's prion *Pachyptila salvini,* blue petrel *Halobaena caerulea*, great-winged petrel *Pterodroma macroptera*, white-chinned petrel *Procellaria aequinoctialis* and grey petrel *Procellaria cinerea.*

Changes in burrow densities and distribution

White-chinned petrels

White-chinned petrels showed the greatest increase in the number of burrows (3.3 times the number of burrows in 1979, Table 3.1, Appendix 3.3). Mike Schramm perceived a visible increase in the number of burrows in 2012 since he was last on the island in 1980. Burrow densities increased most in the steep vegetated slopes (14.6 to 56.1 burrows·ha^{-1}) and coastal lowland habitats (24.6 to 85.6 burrows·ha^{-1}, Fig. 3.5 and Appendix 3.2), with the highest mean density of 113 burrows·ha^{-1} recorded in the Van den Boogaard sample quadrats. Previously *Poa cookii* tussock supported the highest densities of white-chinned petrel burrows (30 burrows·ha^{-1}), but in 2013 both open fernbrake (61 burrows·ha^{-1}) and *Acaena magellanica* herbfield (50 burrows·ha^{-1}) supported higher densities (Appendix 3.1).

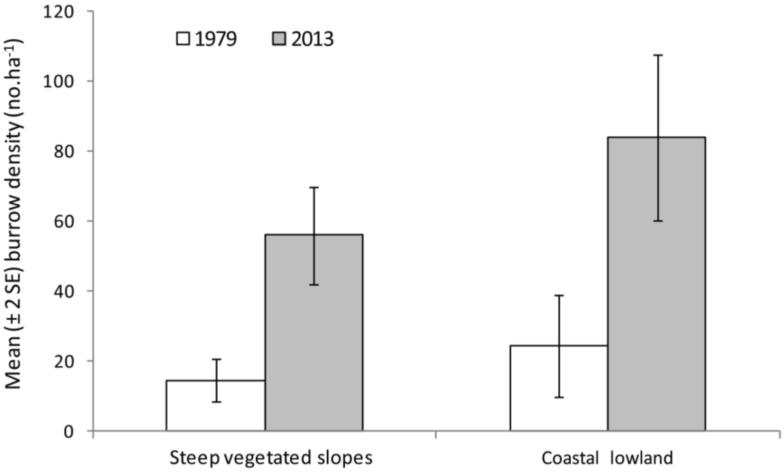


Figure 3.5. Increases in estimated densities (± 95% CI) of white-chinned petrel *Procellaria aequinoctialis* burrow densities in two habitat types in northeast Marion Island from 1979 and 2013.

Salvin's prions

Salvin's prions also showed an overall increase in burrow densities (1.4 times the number of burrows in 1979, Table 3.1 and Appendix 3.2), with the most notable increases in the steep vegetated slopes (64 to 100 burrows·ha^{-1}) and the partly vegetated lava hummocks in Hoppie's Hell (279 to 393 burrows·ha^{-1}). The latter area has become more extensively vegetated since 1979, and now represents more of a 'vegetated lava hummocks' habitat with deeper soils and more vegetation cover for burrows (Niek Gremmen and Valdon Smith, pers. comm. 2013). Salvin's prions showed a strong preference across all study sites for burrowing in *Acaena* (increase from 100 to 211 burrows·ha^{-1}, Table 3.2). There was also an increase in Salvin's prion burrows on the cinder slopes (Table 3.1), particularly the south and east (coastal) facing slopes of Hendrik Vister Kop, where prions utilised lower slopes dominated by *Acaena* and *Azorella selago* (Table 3.3).

Kerguelen petrels

Kerguelen petrels showed little change in their burrow densities. Breeding in early summer, these small-medium sized gadfly petrels prefer steep, vegetated slopes and, to a lesser extent, coastal lowlands, usually choosing a steep slope (mean 35˚) for easy take off. In both surveys the highest densities were recorded on the steep, upper ridges of Blue Petrel Bay. Although their chicks had fledged at time of the survey, the distinctive nature of the burrow, especially the neat and extended

Table 3.1. The estimated numbers of petrel burrows (for eight species) within the six habitat types of the 1041 ha study area at Marion Island. Areas calculate by Schramm (1986).

Habitat type (area)	**Salvin's prion**		**Blue petrel**		**Great-winged petrel**		**Kerguelen petrel**		**Soft-plumaged petrel**		**White-chinned petrel**		**Other petrels**		**All petrels**	
Year	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**
Steep vegetated slopes (71 ha)	4,587	7,157	5,488	8,797	1,243	1,328	518	838	1,271	1,697	1,037	3,983	0	56[2]	14,144	23,899
Fjaeldmark and mire plateaux (147 ha)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Coastal lowland (38 ha)	908	1,733	0	0	84	137	137	110	0	0	935	3196	0	0	2064	5,176
Vegetated lava hummocks (397 ha)	17,826	28,345	596	1,630	14,967	28,008	0	0	0	1,630	1,747	4,927	1151[1]	530[1]	36,287	63,710
Partly vegetated lava hummocks (367 ha)	102,466	144,488	0	0	0	0	0	0	0	771	0	0	770[1]	3,046[2]	103,236	148,305
Cinder slopes (21 ha)	351	920	0	53	80	200	0	0	0	0	8	17	44[2]	16[2]	483	1,208
Total (1041 ha)	126,138	182,643	6,084	10,479	16,374	29,672	655	948	1,271	4,097	3,727	12,123	1,965	3,650	156,214	243,613
% composition	80.7 %	74.8 %	3.9 %	4.3 %	10.5 %	12.2 %	0.4 %	0.4 %	0.8 %	1.7 %	2.4 %	5.0 %	1.3 %	1.5 %	100.0 %	100.0 %
Factor increase in number of burrows		1.44		1.72		1.81		1.45		3.22		3.25		1.86		1.56

[1] Grey petrel
[2] Diving petrel

moat (Appendix 3.1), confirmed their presence along this coastal slope. The transect counts revealed little change in the densities for Kerguelen petrels, but there is a general perception that Kerguelen petrels have become less common over the last few decades. For example Mike Schramm monitored 49 study burrows along the slopes above Gentoo Lake in 1979, but only two active burrows were found in this area in 2012. As a further example of their scarcity, during 26 nights spot-lighting near the Base in October-November 2012 (peak incubation period for this species), only two Kerguelen petrels were sighted in flight.

Soft-plumaged petrels

Although the results from this repeat survey show the number of soft-plumaged petrel burrows have increased 3.2 times since 1979 (Table 3.1), these data are from a small base. This large factor increase is being driven by burrows found in areas which have become more vegetated since the 1979 survey, for example the vegetated lava hummocks on the side of Junior's Kop and the partly vegetated lava hummocks of Hoppies Hell. Moderate increases in the number of burrows in the steep vegetated slopes along the coastal ridges at Albatross Lakes, Blue Petrel Bay and Skua Ridge, and in the steep slopes below the east facing cliffs of Piew Crags all contributed to the overall increase since 1979 (Appendix 3.2). The steep coastal vegetated slopes of Macaroni Bay used to support numerous soft-plumaged petrel burrows, but these slopes are now dominated by dense patches of invasive *Agrostis castellana* and *A. stolonifera* (Gremmen 1997) forming an impenetrable mat of roots and grass unsuitable for burrowing birds. In contrast to the Kerguelen petrels, soft-plumaged petrels are regularly seen and heard at night on Marion, which does lend support to this apparent increase in burrow numbers. However, the overall density of soft-plumaged petrels is still relatively low.

Table 3.2. Mean burrow densities (burrows·ha^{-1}) ± SD of six petrel species in eight vegetation types at Marion Island in 2013.

Vegetation type	Salvin's prion	Blue petrel	Great-winged petrel	Kerguelen petrel	Soft-plumaged petrel	White-chinned petrel	Other petrels	All petrels
Cotula herbfield	0	800.0 ± 707.1	0	16.7 ± 40.8	33.3 ± 51.6	183.3 ± 240.1	0	1033.3 ± 981.2
Poa tussock	75.7 ± 222.9	527.0 ± 929.1	0	8.1 ± 27.7	16.2 ± 44.2	45.9 ± 114.5	2.7 ± 16.4 [2]	675.7 ± 906.0
Closed fernbrake	113.1 ± 204.1	4.9 ± 28.3	41.0 ± 88.8	8.2 ± 30.4	13.1 ± 56.0	20.5 ± 52.9	2.5 ± 15.6 [2]	203.3 ± 234.9
Open fernbrake	98.4 ± 170.5	4.1 ± 34.8	15.9 ± 55.3	4.9 ± 23.4	7.7 ± 37.0	61.4 ± 118.2	0.4 ± 6.4 [1]	192.7 ± 237.9
Acaena herbfield	211.0 ± 236.9	60.4 ± 255.1	39.6 ± 95.3	7.7 ± 34.1	22.0 ± 57.4	50.5 ± 100.4	2.2 ± 14.7 [2]	393.4 ± 343.8
Agrostis mire	18.6 ± 62.7	0	0	0	0	34.9 ± 113.1	0	53.5 ± 162.3
Azorella fjaeldmark	42.9 ± 105.6	0	0	0	0	0	1.8 ± 13.6 [2]	44.8 ± 105.7
Unvegetated	0	0	0	0	0	0	0	0

[1] Grey petrel

[2] Diving petrel

Table 3.3. Mean burrow densities (burrows·ha^{-1}) ± SD of six petrel species in eight vegetation types at Marion Island in 2013.

Vegetation type	**Salvin's prion**	**Blue petrel**	**Great-winged petrel**	**Kerguelen petrel**	**Soft-plumaged petrel**	**White-chinned petrel**	**Other petrels**	**All petrels**
Cotula herbfield	0	800.0 ± 707.1	0	16.7 ± 40.8	33.3 ± 51.6	183.3 ± 240.1	0	1033.3 ± 981.2
Poa tussock	75.7 ± 222.9	527.0 ± 929.1	0	8.1 ± 27.7	16.2 ± 44.2	45.9 ± 114.5	2.7 ± 16.4 [2]	675.7 ± 906.0
Closed fernbrake	113.1 ± 204.1	4.9 ± 28.3	41.0 ± 88.8	8.2 ± 30.4	13.1 ± 56.0	20.5 ± 52.9	2.5 ± 15.6 [2]	203.3 ± 234.9
Open fernbrake	98.4 ± 170.5	4.1 ± 34.8	15.9 ± 55.3	4.9 ± 23.4	7.7 ± 37.0	61.4 ± 118.2	0.4 ± 6.4 [1]	192.7 ± 237.9
Acaena herbfield	211.0 ± 236.9	60.4 ± 255.1	39.6 ± 95.3	7.7 ± 34.1	22.0 ± 57.4	50.5 ± 100.4	2.2 ± 14.7 [2]	393.4 ± 343.8
Agrostis mire	18.6 ± 62.7	0	0	0	0	34.9 ± 113.1	0	53.5 ± 162.3
Azorella fjaeldmark	42.9 ± 105.6	0	0	0	0	0	1.8 ± 13.6 [2]	44.8 ± 105.7
Unvegetated	0	0	0	0	0	0	0	0

[1] Grey petrel

[2] Diving petrel

Great-winged petrels

Great-winged petrels had marginal increases in burrow densities (1.8 times the number of burrows in 1979, Table 3.1 and Appendix 3.2) and still favour the sheltered slopes of vegetated lava hummocks (e.g. Nellie Humps and the sides of Junior's Kop). Burrows were also recorded in dense *Acaena* on the northern lower slopes of Junior's Kop, with an overall increase from 2 to 39 burrows·ha^{-1} in areas dominated by *Acaena* (Appendix 3.2). Schramm (1986) found the highest densities (45 burrows·ha^{-1}) in *Poa* tussock along Skua Ridge. This site is still favoured (103 burrows·ha^{-1}), but is now dominated by *Acaena* and the low fern *Blechnum penna-marina*. The steep vegetated slopes around Albatross Lakes were good burrowing petrel habitat in 1979, especially for great-winged petrels, with deep soils suited to large burrows. However this habitat has since been invaded by the alien grass *Agrostis stolonifera* (Gremmen *et al*. 1998), which forms a dense, impenetrable mat where few burrows were recorded in 2013.

Blue petrels

Blue petrels also had marginal increases in burrow densities (1.7 times the number of burrows in 1979, Table 3.1 and Appendix 3.2), however these results should be viewed with some caution because blue petrels are colonial breeders and thus the results are very sensitive to shifts in colony boundaries. Transects intersected colonies at Macaroni Bay, where birds still favoured the steep coastal slopes dominated by dense *Acaena* patches, and at Blue Petrel Bay, where they favoured the lower coastal slopes dominated by *Poa* tussock (104 to 527 burrows·ha^{-1}, Appendix 3.2). Although this was the highest burrow density recorded in this survey, it is low compared to blue petrel densities at Prince Edward Island, where densities in *Poa* tussock were 2,600–8,300 burrows·ha^{-1} in 1979 (Schramm 1986). The steep vegetated slopes above Ship's Cove have a high density of the invasive *A. stolonifera* (Gremmen *et al*. 1998), however some blue petrels were found in a few isolated *Acaena* patches on these slopes. The upper slopes of the cove are largely free of *A. stolonifera* and are well utilized by burrowing petrels, including blue petrels, resulting in an overall increase in burrow densities for Ship's Cove.

Diving petrels

Two species of diving petrels *Pelecanoides* spp. breed at the Prince Edward Islands: South Georgian diving petrels are largely confined to cinder slopes, whereas common diving petrels nest on well-vegetated coastal slopes. Both species are scarce at Marion Island; only eight active diving petrel burrows were found in 2013. On the scoria cones, 19 burrows were found, but only two had fresh feathers and guano and were assumed to be South Georgian diving petrel burrows. This represents a modest decrease from 1979 (3.2 to 1.3 burrows·ha^{-1}, Table 3.1), but the sample size is too small to make any firm conclusions about trends. Four common diving petrel burrows were found in well established *Blechnum* slopes in Hoppies Hell. One contained an adult common diving petrel, with no sign of a chick or egg, the other three burrows had signs of recent activity. Although breeding was not confirmed, this is the first record of the species ashore in a burrow since it was assumed to have been extirpated by cats (Ryan and Bester 2008). Four skua nests in Hoppies Hell had numerous diving petrel wings, further indicating their presence in the area (Ryan *et al*. 2009b; although some birds flying to the interior may be caught by skuas, Schramm 1986). Active diving petrel burrows at Piew Crags (n =

1) and Blue Petrel Bay (*n* = 1) could not be identified to species. Diving petrel wings were found in sub-Antarctic skua middens south-east of Blue Petrel Bay, but none were found in two middens below Piew Crags.

Grey petrels

Only 15 grey petrel nests were found during the intensive survey of a ≈300 ha area: one in an earth burrow and 14 in rock caves, giving a rough density of 0.05 burrows·ha^{-1} for this area. Using the transect results, burrow densities appear to have decreased (3.1 to 1.0 burrows·ha^{-1}), albeit from a very low base.

Changes in nest site selection

Overall, the estimated densities of burrowing petrels increased across all five habitat types, although overlapping 95% confidence intervals suggest that the increases are not significant in either of the lava hummock habitats (Fig. 3.6).

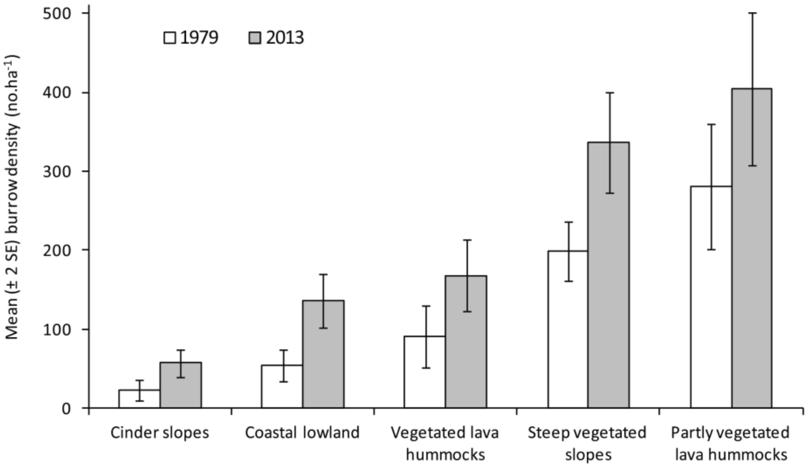


Figure 3.6. Changes in density estimates (± 95% CI) of all petrel burrows (Salvin's prions, blue petrels, soft-plumaged petrels, Kerguelen petrels, great-winged petrels, white-chinned petrels, grey petrels and South Georgian diving petrels) in five habitat types in northeast Marion Island from 1979 and 2013.

Partly vegetated lava hummocks had the highest density of burrows, due to the predominance of Salvin's prions, but the increase in burrow density (44%) was lowest in this habitat (Fig. 3.6, Appendix 3.2). The numbers of burrows in the vegetated lava hummocks increased 69% overall, attributable to the site at the base of Junior's Kop, which increased from 81 to 214 burrows·ha^{-1}, mainly due to increases in Salvin's prions, which are able to utilise shallow soils and natural cavities, as well as species requiring deeper soils such as white-chinned and great-winged petrels (Appendix 3.2). The steep vegetated slopes had the highest burrow densities for blue, great-winged, Kerguelen and soft-plumaged petrels, and showed a significant increase in petrel densities (69%, Fig. 3.6). The large increase in burrow densities in the coastal lowlands (251%) was primarily a result of the increase in white-chinned petrels at Van den Boogaard and Trypot. The cinder slopes also showed a large increase in overall burrow densities (250%), but remained less than half the density in any other habitat type sampled (Fig. 3.6).

The 2013 data confirmed that larger petrel species are associated with sites with deeper soils for their larger burrows (Schramm 1986; Appendix 3.4). Burrow entrances of most species faced mainly to the east, away from the prevailing westerly winds, but also facing downslope towards the sea (Fig. 3.1, Appendix 3.4). Great-winged petrels typically faced more southeast, away from the north-westerly winter winds, differing significantly from the other species surveyed (Appendix 3.4).

Discussion

Burrow densities on Marion Island are still relatively low when compared to neighbouring Prince Edward Island and other Southern Ocean Island groups (e.g. Kerguelen Islands, Barbraud *et al.* 2009; Diego Ramirez Islands, Lawton *et al.* 2006). Burrowing petrel populations are difficult to census precisely (Brooke 2004a), making it hard to conduct comparable repeat counts, especially over long periods. I was fortunate to have continuity in approach between the two studies as Mike Schramm, who made the 1979 counts, was present to ensure that the areas sampled were matched as closely as possible, and to advise on burrow identification and the criteria for discriminating active burrows. The repeat survey was conducted to determine the extent of any recovery of the petrel community following the removal of cats more than two decades ago. Comparison with Prince Edward Island, 22 km north-east of Marion Island, emphasises the impact cats had on Marion's burrowing petrel populations. In 1979, Schramm (1986) estimated that petrel densities on Prince Edward Island were roughly 25 times greater than those on Marion Island, and attributed the difference to predation by the large population of feral cats on Marion (>2,000 cats, Bester *et al*. 2002). If I assume that adult and sub-adult petrels from both islands share similar challenges at sea (e.g. changes in prey availability and distribution (Cherel and Hobson 2007) or fisheries by-catch (Barnes *et al*. 1997)), then the reasons for the huge differences in petrel densities between the two islands are almost certainly due to factors on the islands which affect petrel breeding success and survival. There are no estimates of petrel survival rates, but their breeding success increased immediately following the removal of cats (Cooper *et al.* 1995; Ryan and Bester 2008) and remains at moderate levels (Chapter 5), suggesting that petrel populations have the potential to recover.

When seabirds recolonise an island after extirpation, the growth of a new colony is usually slow and limited, in part, by delayed natal recruitment (Warham 1990). However, with the exception of the common diving petrel (van Aarde 1980), Marion did not experience species extirpations during the cat era and the remaining populations should have had the potential to grow rapidly. At the time of my survey, cats had been absent from Marion Island for 22 years, and their numbers greatly reduced for several years prior to 1990 (Bester *et al*. 2002). Given recovery rates at other islands where introduced predators have been extirpated of around 5–7% per year from endogenous growth alone (e.g. Ryan *et al*. 2006), I would expect petrel numbers to have increased at least 3–5–fold if cats were the sole factor depressing petrel populations. White-chinned and soft-plumaged petrels were the only species to attain these levels of growth. Considering white-chinned petrels are one of the widest ranging of seabirds when breeding (Weimerskirch *et al*. 1999) and are the seabird most often killed on longlines in the Southern Ocean (Delord *et al*. 2010; Petersen *et al*. 2009), their strong recovery relative to other species less prone to fishing mortality is unexpected. The increase in soft-plumaged petrels was driven mainly by their apparent colonisation of vegetated lava hummocks, a habitat from which they were not recorded in 1979 (Table 3.1). Other petrel populations had more modest growth rates over the last few decades, less than doubling their numbers. Thus it appears that while removing the cats was a crucial step towards the recovery of Marion Island's burrowing petrels, other factors may continue to suppress their populations.

In addition to endogenous growth, petrel numbers on islands can also be boosted by immigration. Situated just 22 km to the NE of Marion, Prince Edward Island is an ideal source for petrel immigration to Marion Island. One example is at Aorangi Island where Buller's Shearwaters *Puffinus bulleri* increased at 20% per year with growth enhanced by immigration from nearby (440 m) Tawhiti Rahi Island (Harper 1983). Although the burrowing petrel species at the Prince Edward Islands are probably less aggressive colonisers than Buller's shearwaters and the immigration source is further away, this study shows that immigration allows petrel populations to grow rapidly following the removal of an introduced predator. Under such a scenario, the Marion populations could be expected to have increased more than 40-fold by 2013, to levels similar to those recorded at Prince Edward Island. However, it is possible that the reduced petrel population on Marion Island allowed for an increase in petrel densities on Prince Edward Island through, for example, reduced competition for food resources. The continued slow recovery of the burrowing petrels on Marion could be a result of this density dependent control. Unless Prince Edward Island becomes saturated with birds, to the point where competition for burrow space are limiting population growth, then juveniles and sub-adults are likely to return to their natal breeding grounds (Warham 1990) and not expand to neighbouring Marion Island.

Other seabird colonies have shown marked responses following the eradication of feral cat populations (e.g. Natividad and San Roque Islands off Mexico and Raoul Island off New Zealand, Jones *et al*. 2011). Grey petrels on sub-Antarctic Macquarie Island increased seven fold (8 to 59 burrows in which nesting was confirmed, Schulz *et al*. 2005) within three years of cats being eradicated (year 2000). Since 2003, researchers have seen substantial increases in grey petrels and blue petrels on mainland Macquarie Island, although rates of increase have still to be quantified (Rachael Alderman, pers. comm.). Ascension Island (97 km^2), in the tropical South Atlantic Ocean, once hosted huge seabird colonies, but the introduction of cats in 1815 proved catastrophic for local seabird populations which eventually were restricted to cat-free cliff edges, stacks and islets where their population sizes were limited by nest site availability (Ashmole *et al*. 1994). The eradication of cats in 2003 increased adult survival of sooty terns *Onychoprion fuscatus* and four species of seabird have recolonised the main island from adjacent relict colonies, despite the presence of black rats *Rattus rattus* and house mice (Ratcliffe *et al*. 2010). On Jarvis Island (5 km^2) in the central Pacific Ocean, cats were introduced in 1936, drastically reducing local seabird populations including extirpating numerous smaller species (Rauzon *et al*. 2011). Cats also extirpated both species of introduced rats, but mice survived. By the 1980s cats were eradicated and most of the extirpated seabird species began to recolonise the island. Petrels were much slower to recover than the surface nesting boobies, frigatebirds, noddies and terns, but by 1996 seabird diversity and abundance were returning to historically recorded levels in the presence of mice (cf. Rauzon *et al*. 2011). Why haven't Marion's petrel populations recovered?

Mice are the likely culprit to explain the slow recovery of burrowing petrels at Marion Island. For 30 years the petrel populations were impacted by cats (top-predators) and perhaps mice (mesopredators). While mice probably target eggs and chicks (Fugler *et al*. 1987), reducing reproductive success, cat predation was far more detrimental because they killed chicks and adults, affecting both reproduction and adult survival (Le Corre 2008). Burrowing petrels have long lifespans and low reproductive rates, making their populations very sensitive to changes in adult survival (Warham 1990). Removal of the top-predator benefited adult survival, but may have triggered a 'mesopredator release effect' (Zavaleta *et al*. 2001; Le Corre 2008), whereby rodent numbers expand, increasing their impact on petrel populations (Rayner *et al*. 2007c). However, mice were not an important prey item for cats (van Aarde 1980), so the mouse population may not have been limited by cat predation (van Aarde *et al*. 1996). Mouse densities on Marion Island are thought to be regulated by bottom-up processes; McClelland (2013) reported mouse densities have increased 145% over the past decade due to the local effects of global climate change (warmer, drier and less extreme climate). Peak mouse densities in 2008–11 were 237 mice.ha^{-1} in mire habitats (McClelland 2013), similar to peak densities on Gough Island (266 mice.ha^{-1}; Cuthbert *et al*. 2016), central South Atlantic, where mouse predation has dramatically reduced chick survival rates of burrowing petrels and other birds (Wanless *et al*. 2012; Cuthbert *et al*. 2013a, b; Chapter 4; Appendix 3).

Mouse injured albatross chicks were first recorded on Marion in 2003 and thereafter attacks continued at a low level affecting <1% of the albatross population (Jones and Ryan 2010). In 2015 there was a sudden increase in mice attacks which were widespread across the island affecting 9% of large, well feathered albatross chicks (Chapter 6). In 2016 the frequency and spread of mice attacks was similar to 2015 (Chapter 6). Burrow cameras installed from 2012–2016 revealed that mice frequently enter nest chambers and harass chicks, with fatal attacks recorded on film in the winter months (three grey petrel chicks and one great-winged petrel chick, Chapter 5). It is probable that mice frequently kill burrowing petrel chicks which would account for the lesser recovery of small petrels and winter-breeding petrels at Marion Island (Fig. 3.4), consistent with the patterns detected at Gough Island where mice are significant predators of seabirds (Cuthbert *et al*. 2013a, b; Chapter 4; Appendix 3). Mice on Gough are significantly larger than those on Marion (Cuthbert *et al*. 2016) and although this might confer an advantage in subduing smaller petrel chicks (Chapter 4), recent observations on Marion Island show that large body mass is not necessarily a prerequisite for mice attacking large albatross chicks (Chapter 6).

Shortly after the introduction of cats (1951–52), common diving petrels were regarded by Rand (1954) as being 'common' on Marion. However during 1965–66 van Zinderen Bakker (1971) found no nests and common diving petrels were thought to have been extirpated from Marion (van Aarde 1980). This survey showed a decrease in diving petrel burrow density on Junior's Kop since 1979 (3.2 to 1.3 burrows·ha^{-1}), but an expansion in their distribution (Table 3.1) with burrows now found in the more vegetated slopes of Hoppies Hell, the coastal slopes of Blue Petrel Bay and the vegetated slopes below the eastern cliffs of Piew Crags. No breeding diving petrels were found during this survey, however in

2015 diving petrels were recorded incubating in burrows on the *Poa* slopes of Goodhope Bay (pers. comm. Stefan Schoombie).

Grey and white-chinned petrels have similar habitat requirements for their burrows, preferring deep soils for their large nesting chambers, but grey petrels breed in winter whereas white-chinned petrels breed in summer. Grey petrels are scarce on Marion Island; in 1952, Rand (1954) described them as ‘not common’, so perhaps their numbers were low even before cats arrived (but they are locally common on nearby Prince Edward Island; Ryan and Bester 2008). Mouse predation may be more regular on this species’ chicks, because they hatch in late winter when mice have few other food sources (Gleeson and van Rensburg 1982; Smith *et al*. 2002). By contrast, the white-chinned petrel population has increased more than three-fold since 1979. Its large size coupled with its mid-summer breeding season probably protects its chicks from mouse predation. The population increase at Marion Island contrasts with nearby Ile de la Possession in the Crozet archipelago, where white-chinned petrels are decreasing (Barbraud *et al*. 2008). The wandering albatross *Diomedea exulans* is another species susceptible to longline mortality, but its population is stable at both Marion (Nel *et al*. 2002) and Ile de la Possession (Inchausti and Weimerskirch 2002), but decreasing at South Georgia (Poncet *et al*. 2006). The contrasting fate of the white-chinned petrel population on Marion may reflect the presence of black rats on Ile de la Possession, which are more aggressive predators of petrel chicks than mice.

In summary, densities of burrowing petrels remain low on Marion Island compared to neighbouring Prince Edward Island and other islands in the southwest Indian Ocean which lack introduced mammalian predators. I recommend regular monitoring of burrowing petrels to assess the long-term changes in population size. Annual assessment of breeding success in study colonies of selected species would be valuable to assess the severity of mouse predation. Chicks should be carefully inspected for mouse wounds. Banding adults and fledglings within study colonies should be conducted to assess natal recruitment and adult survival, provided the disturbance does not cause undue emigration from study colonies. An expedition to Prince Edward Island is vital to assess current burrow densities to compare with those estimated in 1979 (Schramm 1986). Eradicating mice from Marion Island would benefit not only the burrowing petrel populations but also help to restore the original structure and functioning of the island’s terrestrial ecosystems. Benefits would be both direct (e.g. recovery of native invertebrate populations, reduced seed predation) and indirect by promoting key ecological processes driven by burrowing petrels (e.g. soil disturbance and marine nutrient imports, especially to inland sites; Caut *et al*. 2012).

Appendix 3.1. Burrow entrance dimensions (mean ± SD, range in parenthesis), physical characteristics and field notes on identifying the burrows of burrow-nesting petrels on Marion Island

Species	Width (mm)	Height (mm)	Wet or dry	Cropped veg.	Nest platform	Field notes
Salvin's prion	112 ± 10 (90-130)[1]	96 ± 9 (80-110)	Dry	No	Flat, simple or none	Fist-sized burrow/rock cavity, minimal nest lining, one entry may lead to multiple chambers
Blue petrel	142 ± 23 (110-210)[1]	110 ± 10 (100-140)	Dry	No	Flat, simple	Narrow passage, usually twists and turns at 90^o, small flat nest bowl
Soft-plumaged petrel	165 ± 24 (140-200)[2]	98 ± 12 (80-120)	Dry (always)	No	Flat, small bowl	Long, dry, narrow burrow entrance, passage to chamber usually straight
Kerguelen petrel	168 ± 23 (140-200)[2]	111 ± 16 (80-140)	Wet	Yes, minimal	Raised, large bowl	'Scaled down white-chinned petrel burrow', can be quite deep, long neat narrow moat
Great-winged petrel	217 ± 50 (140-315)[1]	167 ± 33 (110-230)	Dry (always)	Extensive, 'L' shape	Flat, bowl	Clear 'oblong' patch of cropped vegetation to side of entrance
Grey petrel	238 ± 32 (190-340)[3]	200 ± 29 (140-280)	Dry	Yes, moderate	Flat, large	Burrows or caves utilised, guano stripes outside, large flat nest bowl
White-chinned petrel	269 ± 28 (210-330)[1]	207 ± 31 (150-300)	Wet or dry	Yes, moderate	Raised, very large	Huge, well kept nest chamber, some (~1/3) have entrance moat (especially coastal sites)

Data sources: [1] This study; [2] Schramm (1983); [3] Ben Dilley, FitzPatrick Inst. unpubl. data, Gough Island 2014

Appendix 3.2. Burrow densities (burrows·ha^{-1}) of eight petrel species at 13 sample sites at Marion Island. Total values per habitat type represented as the mean (burrows·ha^{-1}) ± SD

Habitat type & sample site	**Salvin's prion**		**Blue petrel**		**Great-winged petrel**		**Kerguelen petrel**		**Soft-plumaged petrel**		**White-chinned petrel**		**Other petrels**		**All Petrels**		**No. Plots (0.01 ha)**	
Year	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**	**1979**	**2013**
Steep veg. slopes	64.6 ± 130.4	100.8 ± 174.2	77.3 ± 254.4	123.9 ± 452.7	17.5 ± 71.8	18.7 ± 68.6	7.3 ± 30.0	11.8 ± 37.0	17.9 ± 55.6	23.9 ± 63.5	14.6 ± 48.0	56.1 ± 109.6	0.0	0.8 ± 8.9 [2]	199.2 ± 289.9	336.6 ± 499.6	246	246
Blue Petrel Bay	8.9	28.9 ± 58.8	277.8	520.0 ± 920.1	8.9	13.3 ± 45.7	20.0	33.3 ± 60.3	20.0	37.8 ± 68.3	6.7	91.1 ±147.4	0.0	2.2 ±14.9 [2]	342.3	726.7 ± 906.8	45	45
Ship's Cove	66.7	127.3 ± 167.8	6.1	15.2 ± 101.1	4.5	7.6 ± 31.9	6.1	9.1 ± 33.8	16.7	13.6 ± 42.5	19.7	60.6 ± 109.3	0.0	0.0	119.8	233.3 ± 280.8	66	66
Skua Ridge	33.3	66.7 ± 106.1	10.1	0.0	96.7	103.3 ± 154.2	0.0	0.0	3.3	10.0 ± 30.5	23.3	36.7 ± 80.9	0.0	0.0	116.6	216.7 ± 247.8	30	30
Macaroni Bay	103.0	139.4 ± 263.3	169.7	184.8 ± 293.8	0.0	0.0	12.1	9.1 ± 29.1	48.5	45.5 ± 86.9	15.2	63.6 ± 96.2	0.0	0.0	348.5	445.5 ± 380.8	33	33
Albatross Lakes	14.8	88.9 ± 247.0	7.4	0.0	25.9	14.8 ± 45.6	0.0	0.0	0.0	3.7 ± 19.2	29.6	63.0 ± 121.3	0.0	0.0	77.7	170.4 ± 265.7	27	27
Piew Crags	140.0	135.6 ± 143.2	0.0	0.0	0.0	0.0	2.2	11.1 ± 31.7	15.6	31.1 ± 87.4	0.0	17.8 ± 68.3	0.0	2.2 ± 14.9 [2]	157.8	197.8 ± 192.5	45	45
Coastal lowland	23.9 ± 72.8	45.6 ± 104.0	0.0	0.0	2.2 ± 18.9	3.6 ± 18.7	3.6 ± 18.7	2.9 ± 16.8	0.0	0.0	24.6 ± 85.6	84.1 ± 139.5	0.0	0.0	54.4 ± 118.0	136.2 ± 201.4	138	138
Van den Boogaard	25.0	71.7 ± 134.1	0.0	0.0	5.0	8.3 ± 27.9	3.3	5.0 ± 21.9	0.0	0.0	28.3	113.3 ± 151.2	0.0	0.0	61.7	198.3 ± 234.7	60	60
Trypot	23.1	25.6 ± 67.3	0.0	0.0	0.0	0.0	3.8	1.3 ± 11.3	0.0	0.0	21.8	61.5 ± 126.1	0.0	0.0	48.7	88.5 ± 157.1	78	78
Veg lava hummocks	44.9 ± 137.8	71.4 ± 104.6	1.5 ± 12.0	4.3 ± 36.1	37.7 ± 83.6	73.9 ± 94.9	0.0	0.0	0.0	4.3 ± 26.7	4.4 ± 20.4	13.0 ± 41.7	2.9 ± 16.8 [1]	1.4 ± 12.0 [1]	91.4 ± 164.0	168.1 ± 188.2	69	69
Nellie Humps	62.5	75.0 ± 108.2	0.0	0.0	29.2	68.8 ± 92.6	0.0	0.0	0.0	0.0	0.0	2.1 ± 14.4	4.2	2.1 ± 14.4 [1]	95.9	147.9 ± 176.2	48	48
Junior's Side	4.8	61.9 ± 97.3	4.8	14.3 ± 65.5	57.1	85.7 ± 101.4	0.0	0.0	0.0	14.3 ± 47.8	14.3	38.1 ± 66.9	0.0	0.0	81.0	214.3 ± 210.4	21	21
Partly veg. lava hum.	279.2 ± 274.6	393.7 ± 334.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1 ± 14.4	0.0	0.0	2.1 ± 14.3 [1]	8.3 ± 27.9 [2]	281.3 ± 275.9	404.1 ± 335.8	48	48
Hoppie's Hell	279.2	393.8 ± 334.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1 ± 14.4	0.0	0.0	2.1	8.3 ± 27.9	281.3	404.2 ± 335.8	48	48
Cinder Slopes	16.7 ± 89.3	43.8 ± 108.1	0.0	2.5 ± 28.8	3.8 ± 32.1	9.5 ± 51.2	0.0	0.0	0.0	0.0	0.4 ± 6.4	0.8 ± 9.1	2.1 ± 17.0 [2]	0.8 ± 9.1 [2]	22.9 ± 100.5	57.5 ± 134.8	240	240
Junior's Kop	5.1	16.7 ± 53.1	0.0	3.8 ± 35.7	5.8	14.1 ± 62.6	0.0	0.0	0.0	0.0	0.6	1.3 ± 11.3	3.2	1.3 ± 11.3	14.7	37.2 ± 116.5	156	156
Hendrik Vister Kop	38.1	94.0 ± 156.3	0.0	0.0	0.0	1.2 ± 10.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	38.1	95.2 ± 157.5	84	84
																	741	741

[1] Grey petrel

[2] Diving petrel

Appendix 3.3. The total number of active and recently active burrows of eight petrel species at 13 sample sites (total of 741 quadrats of 10x10 m) at Marion Island in 2013. Values under each species are presented as the 'total number of active burrows ± SD' and the 'number quadrats where burrows of this species were found' in parenthesis

Sample site (#quadrats)	**Salvin's Prion**	**Blue petrel**	**Great-winged petrel**	**Kerguelen petrel**	**Soft-plumaged petrel**	**White-chinned petrel**	**Grey petrel**	**Diving petrel**	**All petrels**	**Empty quadrats**
Blue Petrel Bay (45)	13 ± 0.6 (10)	234 ± 9.2 (22)	6 ± 0.5 (4)	15 ± 0.6 (12)	17 ± 0.7 (12)	41 ± 1.5 (17)	0	1 ± 0.1 (1)	327 ± 9.1 (39)	6
Ship's Cove (66)	84 ± 1.7 (36)	10 ± 1.0 (2)	5 ± 0.3 (4)	6 ± 0.3 (5)	9 ± 0.4 (7)	40 ± 1.1 (20)	0	0	154 ± 2.8 (41)	25
Skua Ridge (30)	20 ± 1.1 (11)	0	31 ± 1.5 (14)	0	3 ± 0.3 (3)	11 ± 0.8 (8)	0	0	65 ± 2.5 (18)	12
Macaroni Bay (33)	46 ± 2.6 (13)	61 ± 2.9 (14)	0	3 ± 0.3 (3)	15 ± 0.9 (9)	21 ± 0.9 (12)	0	0	146 ± 3.8 (28)	5
Albatross Lakes (27)	24 ± 2.5 (5)	0	4 ± 0.5 (3)	0	1 ± 0.2 (1)	17 ± 1.2 (8)	0	0	46 ± 2.7 (11)	16
Piew Crags (45)	61 ± 1.4 (27)	0	0	5 ± 0.3 (5)	14 ± 0.9 (6)	8 ± 0.7 (4)	0	1 ± 0.1 (1)	89 ± 1.9 (30)	15
Van den Boogaard (60)	43 ± 1.3 (20)	0	5 ± 0.3 (5)	3 ± 0.2 (3)	0	68 ± 1.5 (30)	0	0	119 ± 2.3 (37)	23
Trypot (78)	20 ± 0.7 (11)	0	0	1 ± 0.1 (1)	0	48 ± 1.3 (24)	0	0	69 ± 1.6 (27)	51
Nellie Humps (48)	36 ± 1.1 (19)	0	33 ± 0.9 (19)	0	0	1 ± 0.1 (1)	1 ± 0.1 (1)	0	71 ± 1.7 (23)	25
Junior's Side (21)	13 ± 0.9 (7)	3 ± 0.6 (1)	18 ± 1.0 (10)	0	3 ± 0.5 (2)	8 ± 0.7 (6)	0	0	45 ± 2.1 (13)	8
Hoppie's Hell (48)	189 ± 3.3 (41)	0	0	0	1 ± 0.1 (1)	0	0	4 ± 0.3 (4)	194 ± 3.4 (41)	7
Junior's Kop (156)	26 ± 0.5 (18)	6 ± 0.4 (2)	22 ± 0.6 (11)	0	0	2 ± 0.1 (2)	0	2 ± 0.1 (2)	58 ± 1.2 (24)	132
Hendrik Vister (84)	79 ± 1.6 (29)	0	1 ± 0.1 (1)	0	0	0	0	0	80 ± 1.6 (29)	55

Appendix 3.4. Mean soil depth (mm), slope (degrees) and aspect (degrees) at nest sites of six petrel species at Marion Island in 2013. Letters connect means not significantly different (*p* > 0.05)

Species	**Soil depth (mm)**	**Slope (degrees)**	**Aspect (degrees)**
Salvin's prion	491 (C)	22.7 (C)	97.5 (B)
Blue petrel	620 (B)	33.2 (A)	90.1 (B)
Soft-plumaged petrel	726 (A, B)	36.4 (A)	87.6 (B)
Kerguelen petrel	700 (A, B)	35.8 (A)	95.7 (B)
Great-winged petrel	702 (A, B)	21.8 (C)	125.6 (A)
White-chinned petrel	795 (A)	23.8 (B, C)	95.1 (B)
All petrels	606 (B)	23.2 (B)	97.6 (B)

Chapter 4

The effects of mouse predation on burrow-nesting petrel chicks at Gough Island

Introduced house mice *Mus musculus* attacking and killing a 10 day old great shearwater *Ardenna gravis* chick in its burrow on Gough Island in 2013 (photo Ben Dilley).

This chapter is broadly based on this publication:

Dilley, B.J., Davies, D., Bond, A.L. and Ryan, P.G., 2015. Effects of mouse predation on burrowing petrel chicks at Gough Island. *Antarctic Science, 27*(6), pp.543-553.

Author contributions: BJD & PGR planned the field work at the study site; BJD developed the burrow cameras, collected and analysed the data and wrote the draft; DD assisted with fieldwork; PGR assisted with data analyses, data presentation and manuscript edits/preparation; AB contributed to drafts.

Chapter 4: The effects of mouse predation on burrow-nesting petrel chicks at Gough Island

Abstract

Since 2004 there has been mounting evidence of the severe impact of introduced house mice *Mus musculus* killing chicks of burrow-nesting petrels at Gough Island. I monitored seven species of burrow-nesting petrels in 2014 using a combination of infra-red video cameras augmented by burrow-scope nest inspections. All seven camera-monitored Atlantic petrel *Pterodroma incerta* chicks were killed by mice within hours of hatching (average 7.2 ± 4.0 hours) with an 87% chick failure rate (n = 83 hatchlings). Four grey petrel *Procellaria cinerea* chicks were found alive with mouse wounds and 60% of chicks failed (n = 35 hatchlings). Video surveillance revealed one (of 7 nests filmed) fatal attack on a great shearwater *Ardenna gravis* chick and two (of 9) on soft-plumaged petrel *Pterodroma mollis* chicks. Mice killed chicks of the recently discovered summer-breeding MacGillivray's prion *Pachyptila macgillivrayi*, with a chick mortality rate of 82% in 2013/14 and 100% in 2014/15. The closely-related broad-billed prion *P. vittata* breeds in late winter and also had a chick mortality rate of 100% in 2014. The results provide further evidence of the dire situation for seabirds nesting on Gough Island and the urgent need for mouse eradication.

Introduction

Many seabirds breed on remote islands in the absence of mammalian predators and so are particularly vulnerable to predation by introduced mammals such as rats *Rattus* spp. and cats *Felis catus* (Croxall *et al*. 2012). Few oceanic islands have escaped invasion by mammals and invasive rodents are likely responsible for the greatest number of bird extinctions from islands (Howald *et al.* 2007).

Gough Island (40˚82'S, 9˚85'W) in the south Atlantic Ocean is an important breeding ground for 23 species of seabird (several of which are globally threatened). As a UNESCO World Heritage Site it has long been considered one of the world's most important seabird breeding islands (Swales 1965). Gough is home to at least 13 species of burrowing petrels, including virtually the entire global population of the Endangered Atlantic petrel *Pterodroma incerta*. It also hosts significant global populations of the Near-threatened grey petrel *Procellaria cinerea*, great-shearwaters *Ardenna gravis*, little shearwaters *Puffinus assimilis*, soft-plumaged petrel *Pterodroma mollis*, Kerguelen petrel *Aphrodroma brevirostris*, broad-billed prion *Pachyptila vittata*, common diving petrel *Pelecanoides urinatrix* and at least three species of storm petrel: white-faced storm-petrel *Pelagodroma marina,* grey-backed storm petrel *Garrodia nereis* and black-bellied storm petrel *Fregetta tropica*. (Brooke 2004a). Breeding success of these burrowing petrels on Gough Island appears to be very poor, with exceptionally low burrow occupancy and breeding success in four species studied from 2009–2011 (Cuthbert *et al*. 2013a). A second population of prions was recently discovered breeding in the summer on Gough Island, morphologically similar to MacGillivray's prion *P. macgillivrayi* from Amsterdam and St. Paul islands in the temperate Indian Ocean (Ryan *et al*. 2014). Blue petrels *Halobaena caerula* were also found breeding on Gough Island for the first time in 2014 (Ryan *et al*. 2015).

House mice *Mus musculus* were brought to the island by sealers in the 19th century and were thought to have little impact on the island's birds, being regarded as 'probably harmless' (Elliott 1953). The impact of house mice on Gough Island's seabird populations has received particular attention since 2001, when mouse predation was identified as the most probable cause of the high chick mortality of Tristan albatross *Diomedea dabbenena* (Cuthbert and Hilton 2004). Breeding success was less than half that of *Diomedea* spp. albatrosses breeding at other locations (Tickell 2000), including those with introduced rats (Possession Island; Weimerskirch 1992) and mice (Marion Island; Nel *et al.* 2003). Island-wide Tristan albatross chick production fell below 10% for the first time in 2014 and is almost exclusively due to predation by introduced house mice (Wanless *et al*. 2009, Appendix 3). Recent observations of mouse-injured Atlantic yellow-nosed *Thalassarche chlororhynchos* and sooty *Phoebetria fusca* albatross chicks are also a cause for concern (Cuthbert *et al.* 2013b), especially given that both species are listed by the IUCN as Endangered.

The hundreds of thousands of petrels that breed in burrows and caves on Gough Island are also affected by mice (Cuthbert and Hilton 2004, Cuthbert *et al.* 2013a). To date, direct evidence of mouse predation on burrowing petrels has been recorded from Atlantic petrel and great shearwater chicks (Wanless 2007). I hypothesise that mice have a negative impact on all burrow-nesting petrels breeding on Gough Island but that the direct evidence is seldom observed, since when compared to the number of recorded chick failures, relatively few chicks have been observed showing wounds characteristic of mouse attacks (Wanless 2007). It is not known how quickly or frequently mice kill petrel chicks, or if they kill them outright or weaken them to the point where they die from their injuries. Mice are quick to scavenge dead chicks in burrows, leaving few clues to determine the cause of death. By understanding the prevalence, nature and speed of mouse attacks these assumptions and high chick failure rates can be qualified to some extent.

I hypothesise that mice can have far more severe effects on burrow-nesting birds than has previously been recognised. I use an array of cameras to investigate the mechanisms and frequency of mouse predations as I suspect that mice attack and kill burrowing petrel chicks very quickly and throughout the year.

Methods

Fieldwork

Fieldwork was conducted between October 2013 and January 2015, covering two breeding seasons for MacGillivray's prion and one breeding season for the other six study species. I made regular nest checks with a burrow-scope to record breeding success, and installed infra-red video cameras at a sub-sample of burrows to record activity inside the nest chambers. I used video cameras to film activity in the nest chamber from hatching to when the chicks were about half-grown for great shearwaters (Dec-Feb), common diving petrel (Dec-Feb), soft-plumaged petrels (Feb-Apr) and MacGillivray's prions (Jan-

Feb) over the austral summer; and for Atlantic petrels (Aug-Sep) over the winter (see Table 4.1 for details). For grey petrels and broad-billed prions, I performed regular nest checks using a burrow-scope to estimate breeding success and monitored chicks for mouse wounds.

Filming nests with video surveillance cameras

I monitored activity inside active burrows with small digital video recorders (DVRs) which connected through video cables (limited to a 200 m range) to a central hub and power supply (constant 220v) in the Meteorological Station at Transvaal Bay. There were sufficient active great-shearwater, soft-plumaged and Atlantic petrel burrows within range of the hub and petrel nests were chosen at random. Each DVR camera (B/W Low Light Mini Camera, code E-25B-B36, 1/3" CCD) was housed in 40 mm PVC piping to keep it dry and secure from mouse damage, fitted with a wide angle lens (2.1 mm Board Lens, covering 120˚), and accompanied by a ring of 12 infra-red light-emitting diodes (LEDs). Inspection hatches were dug through the roof of the burrow passage to gain access to the nest chamber. Each camera was then secured to a metal angle-iron pole and positioned 20–30 cm away from the incubating bird. The eight motion activated cameras connected to a video surveillance system (SuperDVR software) which enabled a live feed to the nearby bird lab and footage was recorded onto a computer.

In addition to the DVR video cameras I also made a weather-proof 'housing' for a digital DSLR camera with a 14 mm fish eye lens to be able to capture high quality colour images of mice attacking burrow-nesting petrels. This DSLR camera was mounted inside the burrow and could be triggered remotely (using a manual shutter release which was extended to reach 200 m from the burrow to the lab) when a mouse was observed inside the burrow on the live video feed. Despite extensive searches in a 200 m radius from the station, only two common diving petrel burrows were found. Both burrows had new nest material in the chamber with an adult present, but only one pair laid an egg and this burrow was monitored with a DVR camera. No storm-petrel nests were found, despite reasonably large numbers of white-faced storm-petrels active around the Station at night.

Cameras were installed into the burrow chamber at roughly mid-incubation, when the occupants would not likely abandon their nest due to the disturbance (Blackmer *et al*. 2004). Camera installation took <10 minutes and did not result in any immediate nest failures. Since I suspected the mice would depredate newly hatched and newly independent chicks, it was important to have the cameras *in situ* before hatching started.

Prions breed in burrows and natural rock cavities, however there were no active prion burrows within range of the DVR camera system. I therefore chose to monitor birds in Prion Cave (40˚ 21.161'S, 9˚ 53.114'W), an accessible cave where MacGillivray's prions breed (Ryan *et al.* 2014; previously reported as broad-billed prions by Cuthbert *et al.* 2013a). Two nests with wounded chicks, found at approximately 28 days old, were filmed with a GoPro camera and an external 12V red light to record mouse-chick interactions.

Breeding success

I monitored Atlantic petrel (n = 92), soft-plumaged petrel (n = 42), and great shearwater (n = 147) burrows (Table 4.2) along established monitoring transects (Cuthbert *et al.* 2013a) and supplemented these with additional nests located closer to the Station. All burrows were checked with a burrow-scope to determine contents, as relying on other cues or responses can overestimate occupancy, and so underestimate breeding success (Rexer-Huber *et al.* 2014). The burrow-scope was custom-made using a high resolution conical pinhole camera, LED torch and a 7" colour monitor, producing a clear image of the inside of the burrow. Burrows were checked during early incubation, and again when chicks were predicted to be hatching and then fledging.

To estimate the breeding success of summer-breeding MacGillivray's prions, I monitored 60 nests in Prion Cave where nests were checked every five days from pre-laying to hatching and then every 10 days until chicks fledged. Broad-billed prions breed in late winter and seven nests in a rock cave at Snoekgat (40˚ 20.88'S, 9˚ 52.72'W), and 11 nests in burrows above the Golden Highway, (40˚ 20.52'S, 9˚ 53.27'W), were checked at mid-incubation and again at the small chick stage, although some early chick failures may have been missed.

Grey Petrels breeding in Gonydale were monitored (n = 41 burrows) from laying to fledging. Burrows were fitted with observation hatches to allow a direct view of the nest chamber with a burrow-scope, and were checked every 4 days from pre-laying to hatching and then every 7–10 days until the chicks fledged.

Relating chick size at hatching to chick survival

Broad-billed prion, MacGillivray's prion and grey petrel eggs were measured (length, L, and maximum breadth, B) to the nearest 0.1 mm using Vernier callipers. The fresh mass of eggs (g) was estimated from the relationship: mass = K_w x LB^2 (Hoyt 1979), where L and B are in cm and the constant $K_w = 0.51$ for all species (Warham 1990). Egg masses for other species were obtained from the literature (Table 4.3). Chick mass at hatching was estimated as ~2/3 of fresh egg mass, as this is typical of petrels (Payne and Prince 1979, Schramm 1983, Booth *et al.* 2000).

Data analysis

The video files recorded a date and time stamp which enabled us to record a detailed sequence of activity for each filmed nest, including hatching date, frequency of mouse attacks, age of the chick when it was first left alone, and the time of death (for chicks that died before fledging). For all species, hatching success was calculated as the proportion of eggs that produced live chicks; this was a maximum estimate as not all eggs were monitored from laying. Fledging success was calculated as the proportion of hatched chicks that survived to fledge, and the total breeding success as the proportion of eggs laid that produced fledged chicks. Since individual mice could not be identified in the footage, the maximum number of mice involved in an attack was recorded as the maximum number of mice in the frame at one time. Regression analyses were conducted in the R 3.1.2 (R Core Team 2014) where I used a binomial generalised linear model (explanatory variable = estimated chick mass at hatching; response variable = chick survival) run in package lme4 (Bates *et al*. 2014). Means are presented ± SD.

Results

Video cameras captured the first confirmed records of fatal attacks by mice on soft-plumaged petrel and MacGivillray's prion chicks. Attacks were also recorded on great shearwater and Atlantic petrel chicks, adding further evidence of attacks on these species since the first records from 2004. In addition, live broad-billed prion and grey petrel chicks were found with wounds typical of those inflicted by mice (see Table 4.3). Video recordings showed the speed with which mice kill chicks and that mice have severe effects on burrow-nesting birds breeding success. My results show that mice affect burrowing petrels year round.

Broad-billed prion

Despite extensive searches, only 18 nests with an incubating bird were located by mid-September 2014: seven nests in Snoekgat cave and 11 nests in burrows on the path to the Golden Highway. By 28 September 2014 all the nests at Snoekgat cave had failed with evidence of mouse incisor marks on freshly broken egg shells (Fig. 4.1) and no evidence of any eggs having hatched. Only 2 of the 11 burrows on the path to the Golden Highway contained chicks by 15 October 2014, and both had failed by 6 November 2014 (18% hatching success and 0% breeding success). On 29 October 2013 a small prion chick (~2 weeks old) was found alive, but with severe mouse wounds, in its burrow on the northeast slopes of 960 Hill (Fig. 4.1).

Figure 4.1. Evidence of mouse incisor marks on freshly broken broad-billed prion egg shell at Snoekgat Cave in September 2014; and a broad-billed prion chick (~2 weeks old) found alive, but with severe mouse wounds, in its burrow on the northeast slopes of 960 Hill, Gough Island, in October 2013 (photos Ben Dilley).

Table 4.1. Summary of the video surveillance results from filming the nests of four petrels on Gough Island in 2014. The motion activated infra-red cameras were installed into the burrow chamber at roughly mid-incubation and connected to a video surveillance system enabling a live feed and recording of footage onto a computer. Values expressed as mean ± SD.

	Pelecanoides urinatrix	***Ardenna gravis***	***Pterodroma mollis***	***Pterodroma incerta***
Number of nests filmed	1	7	9	8
Total days filmed	75	241	288	90
Number of chicks hatched	1	6	7	7
Number of chicks killed by mice	0	1	2	7
Chick failure rate due to mice	0%	17%	29%	100%
Chick age when left alone (days)	10.1	3.9 ± 0.6	1.8 ± 0.4	(all killed)
Chick age when first attacked (days)	11.8	9.4	51.9 ± 13.8	0.11 ± 0.07
Max. number of mice attacking at one time	3	3	1.5 ± 0.7	1.9 ± 0.7
Time from first attack to death (days)	survived	3.3	2.3 ± 0.9	0.19 ± 0.17
Chick age at death (days)	survived	12.7	54.2 ± 12.8	0.30 ± 0.17

Great shearwater

Seven great shearwater nests were filmed for 241 days from mid-late incubation (mid-December) until chicks were medium-sized (mid-February; Table 4.1) when cameras were removed to install into soft-plumaged petrel burrows at late incubation. In one nest, during the last week of incubation, the parent abandoned its egg after 23 days without relief from its partner and 48 minutes later, a mouse entered the burrow and attempted unsuccessfully to gnaw into the egg. At 59 minutes after the adult departed, a larger mouse arrived and made a hole through the pointed end of the egg, and within 74 minutes of the egg being abandoned it had been reduced to a small fragment of egg shell with mouse bite marks. An adult great shearwater (presumably a parent) entered the burrow 130 minutes after the egg was abandoned and settled on the nest mound until joined by another adult two days later, before both birds abandoned the burrow.

The chicks hatched in the remaining six filmed nests, and were left alone after 3.9 ± 0.6 days (range 3.1–4.5 days). One chick was wounded on the lower rump by a single mouse 9.4 days after hatching. Following repeated attacks over 3.3 days by up to three mice at a time, the chick died (Fig. 4.2). When first attacked, this chick appeared in good health and had been fed by a parent on two occasions since being left alone at 4.3 days old. Although the other five chicks were frequently visited and occasionally agitated by mice, none were wounded and all survived to fledge (nests were monitored with a burrow-scope after the cameras were removed). Great shearwater fledgling success was 60% and breeding success was 44% in 2014 (n = 147 nests, Table 4.3).

Figure 4.2. This great shearwater chick (here 10 days old) died 3.3 days after first being attacked by a mouse (photo Ben Dilley).

Table 4.2. Burrowing petrel nests monitored on Gough Island in 2014.

Breeding success	***Pachyptila macgillivrayi***	***Pachyptila vittata***	***Pterodroma mollis***	***Pterodroma incerta***	***Ardenna gravis***	***Procellaria cinerea***
Nests monitored	60	18	42	92	147	41
Eggs hatched	51	2	30	83	106	35
Egg failure rate	15%	89%	28%	14%	28%	14%
Chicks fledged	9	0	19	11	64	14
Chick failure rate	82%	100%	37%	87%	40%	60%
Breeding success	15%	0%	45%	12%	44%	34%

MacGillivray's prion

Eggs were laid in Prion Cave from approximately 23 November to 3 December 2013 (n = 60 nests) and chicks hatched in the first week of January 2014 (51/60, 85% hatching success). Chicks were brooded for 5–10 days after hatching. Chick survival was very low, with 9/51 chicks surviving to fledge in late February 2014 (18%), giving an overall breeding success of 15% (Table 4.2). Almost all of the chick failures (93%, *n* = 42) occurred in the first 10 days of February 2014 when chicks were >20 days old. GoPro footage of an injured chick showed two mice gnawing at its neck wound (Fig. 4.3) with more mice and two Gough moorhens *Gallinula comeri* also feeding off dead chicks in the cave. In December 2014, 60 nests were again monitored in Prion Cave. Further video evidence of mice attacking and killing chicks was recorded and by the first week of February 2015 all chicks had died, giving an average breeding success over both years of 7%.

Figure 4.3. Mice attacking a MacGillivray's prion chick (here ~20 days old) in Prion Cave.

Table 4.3. Summary of the hatching period (listed in succession from spring-summer-winter) and fresh egg mass (see methods for details) in relation to chick survival (2014 data in bold, previous years in parenthesis from Cuthbert *et al*. 2013b) for a selection of burrow nesting petrels and the endemic Gough bunting on Gough Island in 2014. Shaded area indicates winter species.

Species	Hatching period	Egg mass (g) mean ± SD	Egg data source	Chick survival	Predation by mice
Pachyptila vittata	early Oct	34.3 ± 2.4	This study	**0.00** (0.00-0.09)	wounded chicks
Pelagodroma marina	Oct-Nov (assumed)	12.8 ± 0.84	Campos & Granadeiro 1999	unknown	highly likely
Garrodia nereis	Oct-Nov (assumed)	8.5	Ryan 2007	unknown	highly likely
Puffinis assimilis	Oct-Nov (assumed)	40.0 ± 3.9	Booth *et al.* 2000	unknown	unknown
Rowettia goughensis	Nov-Dec	5.4 ± 0.3	Ryan & Moloney 2002	unknown	highly likely
Aphrodroma brevirostris	end Nov-Dec	57.1 ± 3.9	Schramm 1983	unknown	likely
Halobaena caerula	Dec (assumed)	42.0 ± 3.5	Fugler *et al.* 1987	unknown	unknown
Pelecanoides urinatrix	mid-Dec	17.4 ± 2.1	Payne & Prince 1979	unknown	unknown
Ardenna gravis	early Jan	94.4 ± 8.6	Cuthbert 2006	**0.60** (0.56-0.96)	confirmed
Pachyptila macgillivrayi	early Jan	39.1 ± 2.9	This study	**0.09** (mean 2014/15)	confirmed
Pterodroma mollis	mid-Jan-Feb	54.4 ±3.9	Schramm 1983	**0.63** (0.14-0.44)	confirmed
Fregetta spp.	mid-Feb (assumed)	9.7 ± 0.6	Quillfeldt & Peter 2000	unknown	unknown
Procellaria cinerea	late April-early June	126.4 ± 8.9	This study	**0.40** (0.31-0.35)	wounded chicks
Pterodroma macroptera	July (assumed)	80.8 ± 5.9	Schramm 1983	unknown	highly likely
Pterodroma incerta	mid-Aug-late Sept	87.4 ± 8.5	Cuthbert 2004	**0.13** (0.36-0.69)	confirmed

Soft-plumaged petrel

Of the nine soft-plumaged petrel burrows filmed, seven chicks hatched; the other two nests failed when the eggs were left unattended and were eaten by mice. One egg was abandoned by the parent seven days after an incubation shift change and within 16 minutes two mice appeared in the burrow and ate the egg, which appeared to contain a well developed chick. This burrow remained empty for a further four nights before an adult occupied the burrow overnight. The other egg was left alone four days after an incubation shift change and was eaten by a single mouse after 3.1 hours. Two days later an adult returned to the burrow. The seven chicks that hatched were left alone after 1.8 ± 0.4 days (range 1.4–2.3 days), and these small chicks appeared to be extremely vulnerable to mouse predation (being of a similar body size to an adult mouse). All seven chicks were frequently visited and agitated by mice, which appeared to lick the chicks' down, presumably feeding off food spilt when the parents fed their chick. However, both fatal chick attacks by mice occurred in April when chicks were much larger (age 42 and 61 days, Table 4.1). No wounds were seen on the five chicks that survived to fledge. The chick survival rate in 2014 was 63% (n = 42 nests), with a 45% breeding success (Table 4.2).

Grey petrel

Grey petrels breeding in Gonydale were checked from laying to fledging (n = 41 burrows). Hatching success was 85% and 14/35 chicks survived to fledge (40%). Overall breeding success was 34% (Table 4.2). Four grey petrel chicks were found alive with mouse wounds on the lower rump (Fig. 4.4), of which three were dead within a week and one survived. For the remaining chick failures, three were killed by brown skuas *Stercorarius antarcticus* that dug up their burrows, three died from unknown causes and 12 (57%) were found dead and partly mouse-eaten in their burrows. These 12 chicks were almost certainly killed by mice, as in all cases the chicks appeared in good health on the previous visit 7–10 days prior. It is therefore likely that mice were responsible for 71% (15/21) of the chick failures.

Figure 4.4. Grey petrel chick (2 weeks old) with mouse injury (photo Ben Dilley).

Atlantic petrel

Cameras were placed in eight Atlantic petrel burrows at late incubation and monitored for a total of 90 days. One egg was abandoned on 24 September 2014 after prolonged incubation and when inspected, the egg was found to be addled. The other seven eggs hatched between 23 August and 13 September. All seven chicks were attacked by mice within 2.7 ± 1.7 hours of hatching (range 0–4.8 hours) and were killed by mice within 7.2 ± 4.0 hours of hatching (range 3.1–15.1 hours, Table 4.4). In all cases chicks were still being brooded and the initial attack was by a single mouse, which was not deterred by the presence of the adult petrel. The mouse would grasp the chick with its front feet while standing on its hind legs and gnaw at one spot until the chicks' skin was broken (Fig. 4.5).

Figure 4.5. Mouse attacking a newly hatched Atlantic petrel chick with the parent sitting alongside (photo Ben Dilley).

Mice appeared to attack whichever part of the small chick was exposed, starting with the rump (4 cases), top of the head (2 cases) or back of the neck (1 case). Once weakened, the mice would expose a large wound on the lower rump, characteristic of wounds seen on freshly dead chicks commonly found in burrows by fieldworkers since 2004. On average 1–3 mice would attack at one time (mean 1.9 ± 0.7 mice) and kill the chick within 4.5 ± 4.0 hours (range 1.5–13.1 hours; Fig. 4.6). In one nest, the chick was just hatching when a mouse pulled off the cracked eggshell, attacked the wet chick, and killed it within 4 hours. Some adults dropped their wings to better cover the newly hatched chick, but the mice pushed underneath the wing, eventually causing the adult to move aside. All attacks were initiated at night, but mice did return to kill injured chicks during the day at two nests (Table 4.4). A one minute video of an Atlantic petrel chick being attacked by a mouse is available at http://youtu.be/VVehgRcfO98. Hatching success was 90% at 92 monitored burrows, but chick survival was 13% (11/83). The timing of chick failures followed a similar pattern to the camera monitored nests, with most failures occurring shortly after hatching.

Table 4.4. The speed with which mice killed seven newly hatched Atlantic petrel chicks. The dashed line (-) represents an egg being incubated; 'H' indicates hatching; the numbers following 'H' represent the hourly maximum number of mice attacking a chick at one time; 'X' indicates a dead chick. Shading represents hours of darkness.

Nest	**18h00**					**Night**								**06h00**				**Day**				**17h00**	
1	-	-	-	-	-	-	H	0	1	1	1	0	0	0	0	0	0	0	0	1	1	2	**X**
2	-	H	1	1	1	**X**																	
3	H	0	0	0	0	1	3	**X**															
4	-	-	-	-	-	-	-	-	-	-	H	2	2	1	1	**X**							
5	-	H	0	0	0	1	1	0	1	2	**X**												
6	-	-	-	H	0	0	1	1	2	2	**X**												
7	-	-	-	-	H	0	0	0	1	1	**X**												

Figure 4.6. A sequence of photos taken inside an Atlantic petrel burrow on Gough Island on evening of the 8th and early morning of the 9th of September 2014 (Nest 5; see Table 4.4 above). In the first photo the adult is shifting around as the chick is hatching; before the chick has even emerged from the shell a mouse (indicated by the red line) is investigating (20h23); mice feed off the egg shell remains (from 22h00) before removing the egg shell from the nest bowl (23h01); the first mouse attacks start from 23h30 (not pictured here) and by 05h00 the next morning the chick is dead (photos Ben Dilley).

Factors affecting chick survival

Although there was a trend for larger chicks to have higher survival (Fig. 4.7), the relationship was not significant ($r^2 = 0.198$, $F_{1,4} = 0.991$, $P = 0.376$) largely due to high chick survival of soft-plumaged petrels. The likelihood of a chick surviving to fledge appears to be related to the time of hatching (season) and its mass at hatching, with both prion species having the lowest chick survival rates and all winter breeders having low chick survival rates.

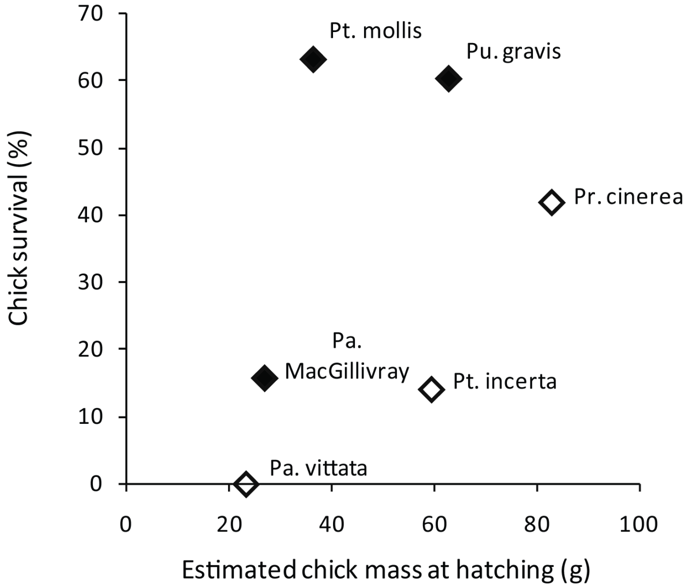


Figure 4.7. Relationship between the estimated chick mass at hatching (g) and the chick survival (%) for burrowing petrels in 2014–2015. Solid diamonds indicate summer breeders and open diamonds indicate winter breeders. Species abbreviations are (from top): soft-plumaged *Pterodroma mollis*, great-shearwaters *Ardenna gravis*, grey petrel *Procellaria cinerea*, MacGillivray's prion *Pachyptila macgillivrayi*, Atlantic petrel *Pterodroma incerta*, broad-billed prion *Pachyptila vittata*.

Discussion

This study shows that mice kill chicks of all species of burrowing petrels studied on Gough Island. The impact of mice on chicks of surface-nesting albatrosses has been well documented, as these species are readily observed and are therefore easier to monitor (Cuthbert and Hilton 2004; Wanless *et al.* 2009; Appendix 3). In 2004, video cameras recorded fatal attacks by mice on burrowing petrel chicks (Wanless 2007), but since 2004 there have been few direct records of mouse interactions with burrow-nesting petrels because of the technical difficulties in observing inside burrows (Brooke 2004a). Once killed, a chick carcass is usually completely consumed by mice or removed from the burrow by moorhens, leaving little evidence as to the reason for the nest failure, or if the chick even hatched. This study showed Atlantic petrel chicks were killed within hours of hatching and the carcass was consumed quickly, which explains why so few mouse-injured chicks have been found during routine nest checks relative to the total number of chick failures. Atlantic petrel breeding success in 2014 was lower than any other year monitored to date (range 36–69%, Wanless *et al*. 2012; Cuthbert *et al*. 2013a). With the exception of one mouse-injured grey petrel chick which recovered, all mouse injured chicks died from their injuries.

In winter mice have limited food resources (Cuthbert *et al.* 2016) and the winter breeding petrels were worst affected by mice, with chicks hatching in early winter (grey petrels) having a higher chick survival rate than chicks hatching in mid winter (Atlantic petrels) or late winter (broad-billed prions, Fig. 4.7). Other winter-breeding species have not been studied because of difficulty locating their burrows, but late winter breeders such as little shearwaters and great-winged petrels *Pterodroma macroptera* (Table 4.3) are probably also severely affected by mouse predation. Little shearwaters have become rare around the Station on Gough over the last 30 years (Peter Ryan pers. comm.), and field workers have been unable to locate any great-winged petrels. In 1955, great-winged petrels occurred in 'large numbers' and little shearwaters were 'extremely abundant' around The Glen on the east coast of the island (Swales 1965).

Previous research has shown smaller seabirds are more vulnerable to rodent predation (Jones *et al*. 2008) and my inability to locate any storm-petrel nests suggests that small species have higher mortalities on Gough and their populations are greatly reduced. Video footage from inside the 25 monitored burrows (Table 4.1) showed that incubating birds often left their burrows for a short period (<10 minutes), usually in the early evening, and some individuals more frequently than others. In some cases birds were absent for a few hours or even days, allowing mice to eat their egg (Fig. 4.8). Temporary egg desertion has been documented for many procellariiforms, and eggs may still hatch despite being neglected for up to two days (Boersma and Wheelwright 1979). Campos and Granadeiro (1999) recorded white-faced storm-petrels on Selvagem Grande Island leaving their eggs for 1–6 days, resulting in 17/35 (48.6%) eggs failing due to mice predation, 12 of which were eaten by mice within 24 hours of the being left alone. Temporary egg desertion is also frequent in blue petrels (Ancel *et al*. 1998), a species recently found breeding on Gough Island in the summer of 2014 (Ryan *et al.* 2015).

Figure 4.8. Mouse predating on a temporarily neglected great shearwater egg (photo Ben Dilley).

Gough Island mice are 50–60% heavier than those from any other island (mean: 35 g; Cuthbert *et al.* 2016). Peak mouse densities are also among the highest recorded for island populations (266 mice.ha^{-1}) with relatively low seasonal variations (4–5 fold) driven primarily by an absence of other mammalian predators and an abundance of seabird chicks as a winter food source (Cuthbert *et al.* 2016). Larger mice are better able to bite into seabird eggs, and presumably have an advantage when attacking seabird chicks. Given that chicks of the two largest burrowing petrels on Gough, grey petrel and great shearwater, are both killed, it is likely that all species are impacted.

Mice have been present on Gough Island for more than a century, so how have these petrels managed to maintain their populations in the face of this predation? Firstly, it is likely that predation may not have been constant, but rather a relatively new behaviour driven in part by environmental change (for more details on this see Chapter 7). Secondly, apart from 1957, there are few detailed records on Gough's seabirds prior to 2000 when year-round seabird monitoring and research began, therefore it's not known how long mice have been affecting chick survival. There are few early records on burrowing petrel populations, but Tristan albatross are better documented and attacks on chicks have almost certainly been happening since at least the 1970s and have probably contributed significantly to an estimated 50% decrease in the breeding population over 50 years (cf. Wanless *et al*. 2009). Lastly, mice target eggs and chicks, reducing petrel reproductive success, but adult survival is not directly affected and this is more important for maintaining their populations (Le Corre 2008). Although burrowing petrels have long lifespans and low reproductive rates, some natal recruitment would be required to maintain adult populations. The MacGillivray's prion and blue petrel have recently been discovered breeding on Gough, but it's not known if these populations were overlooked or if they recently colonised the island.

Elsewhere, introduced rodents coexist with burrow-nesting seabirds, but seabird populations are either supplemented by immigration from other colonies, or predation is lower due to smaller rodent populations (Quillfeldt *et al*. 2008; Brooke *et al*. 2010). In other cases, however, seabird populations are too large to census accurately, and the effects of introduced rodents may not be immediately recognized (Major *et al*. 2013). The nearest potential sources of immigrants to Gough are Inaccessible and Nightingale islands, 400 km away (Ryan 2007), making inter-island movements of petrels unlikely

(Brooke 2004b; Buxton *et al*. 2014). I believe that given the relatively recent arrival of mice (<200 years), their strong initial bottleneck (Gray *et al*. 2014), the generally high adult survival and longevity of petrels (Brooke 2004a), and initially large populations of petrels (Swales 1965) have all combined to result in the persistence of petrels on Gough. Their continued persistence, though, is perilous in the face of the intense mouse predation I documented.

My estimates of breeding success in 2014 were similar to or higher than recent estimates for summer-hatching species (Cuthbert *et al.* 2013a), but in 2014 Atlantic petrels had the lowest breeding success yet recorded, as was the case for Tristan albatrosses (<10%, Appendix 3). The low breeding success of Atlantic petrels is of particular concern since virtually the entire population breeds on Gough Island. This species has not been recorded breeding on the main island of Tristan da Cunha for 40 years and is probably extinct there (Ryan 2007), but small numbers may breed on Inaccessible Island (Peter Ryan pers. comm.). Of equal concern is the recently discovered population of MacGillivray's prion on Gough Island (Ryan *et al*. 2014). This species is extinct on Amsterdam Island, and only a relict population of at most a few hundred birds breeds on La Quille, a stack off St Paul Island (Worthy and Jouventin 1999). If, as seems likely, the Gough population is part of this species (Peter Ryan, unpublished data), Gough supports virtually the entire world population. MacGillivray's prion chick survival in Prion Cave was low in both 2014 (18%) and 2015 (0%). This is much lower than the 60–70% chick survival rate by prions breeding at predator-free islands (Liddle 1994).

Broad-billed prions had the worst breeding success of all species monitored in 2014. In addition, a substantial search effort was needed to find burrows containing incubating adults, despite their being the most common petrel seen at night around the Station. Most nests failed at the egg or early chick stage (16/18 eggs laid) and the only two small chicks recorded also disappeared, resulting in 0% breeding success. These results are similar to previous years when small samples of nests gave breeding success estimates of 0–9% (Cuthbert *et al.* 2013a).

My study therefore confirms that house mice are significant predators of petrel eggs and chicks on Gough Island, and that all species are likely to be impacted. Video footage showed that mice can be very effective predators of burrowing petrels, killing chicks within hours of hatching while still brooded by their parents, and also tackling large chicks of species many times their body size. Gough Island is the highest priority island for introduced vertebrate eradication in the UK Overseas Territories (Dawson *et al.* 2015) and urgent action is needed if prospects for seabirds on Gough Island are to be improved. Petrels, particularly the smaller and rarer species, are likely to be extirpated from Gough if mice are not eradicated in the near future. Preparations for such an operation are complex, but are ongoing (Broome and Garden 2013; see Chapter 7 for an update on the eradication plans).

Chapter 5

Mouse predation affects breeding success of burrow-nesting petrels at sub-Antarctic Marion Island

Inside a small cave on Marion Island (top photo) where a two week old grey petrel *Procellaria cinerea* chick was filmed using an infra-red digital video camera (white pipe) to reveal how mice repeatedly hassled the chick (black and white video snapshots; photos Ben Dilley).

This chapter is broadly based on this publication:

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Chapter 5: Mouse predation affects breeding success of burrow-nesting petrels at sub-Antarctic Marion Island

Abstract

I report the breeding success of four species of burrow-nesting petrels at sub-Antarctic Marion Island where house mice *Mus musculus* are the sole introduced mammal. Feral cats *Felis catus* were present on Marion for four decades from 1949, killing millions of seabirds and greatly reducing petrel populations. Cats were eradicated by 1991, but petrel populations have shown only marginal recoveries. I hypothesize that mice are suppressing their recovery through depredation of petrel eggs and chicks. Breeding success for winter breeders (grey petrels *Procellaria cinerea* (34 ± 21%) and great-winged petrels *Pterodroma macroptera* (52 ± 7%)) were lower than for summer breeders (blue petrels *Halobaena caerulea* (61 ± 6%) and white-chinned petrels *Procellaria aequinoctialis* (59 ± 6%)) and among winter breeders, most chick fatalities were of small chicks up to 14 days old. I assessed the extent of mouse predation by monitoring the inside of 55 burrow chambers with video surveillance cameras (4024 film days from 2012–16) and recorded fatal attacks on grey (3/18 nests filmed, 17%) and great-winged petrel chicks (1/19, 5%). My results show that burrow-nesting petrels are at risk from mouse predation, providing further motivation for the eradication of mice from Marion Island.

Introduction

Burrow-nesting petrels are the most abundant seabirds in the Southern Ocean, with a total population in the hundreds of millions of birds (Warham 1996). Having evolved as insular birds breeding on remote oceanic islands, they lack behavioural adaptations that allow them to coexist with introduced mammalian predators (Blackburn *et al*. 2004). Since few oceanic islands have escaped invasion, introduced predators (e.g. domestic cats *Felis catus*, rats *Rattus* spp. and house mice *Mus musculus*) account for the largest proportion of seabird population declines, more so than incidental bycatch and competition for prey with commercial fisheries (Jones *et al*. 2008).

House mice were introduced accidentally to sub-Antarctic Marion Island (46°54'S, 37°45'E) in the early 19th century, most probably by sealers or shipwrecks (Watkins and Cooper 1986). Domestic cats were taken to the island’s weather station in 1948 to control mice, but they soon turned feral and started eating the island’s seabirds (Rand 1954). By the mid-1970s an estimated 2,000 cats were killing some 450,000 birds per year, most of which were burrow-nesting petrels (Van Aarde 1980). Petrel population densities were reduced more than 20-fold compared to the adjacent, predator-free Prince Edward Island (Schramm 1986), and some small species (e.g. diving petrels and storm petrels) were apparently extirpated (Van Aarde 1980, Ryan and Bester 2008). Fortunately cats were eradicated by

1991 (Bester *et al*. 2002), allowing the greatly diminished burrow-nesting petrel numbers to recover.

Initial indications were positive. Following the removal of cats there were marked increases in the breeding success of burrowing petrels, especially great-winged petrels *Pterodroma macroptera*, which breed in winter when cat predation pressure was most severe (Cooper and Fourie 1991, Cooper *et al*. 1995). However, the post-cat recovery of burrowing petrel numbers on Marion has been much slower than anticipated, especially for smaller species (Chapters 3 and 5). Recent evidence from a repeat survey of burrow densities (Chapter 3) and from analyses of prey remains of brown skuas *Stercorarius antarcticus* (Cerfonteyn and Ryan 2016) both suggest there has been little recovery of burrow-nesting petrel populations at Marion. At least nine species of burrow-nesting petrels breed on Marion Island (Ryan and Bester 2008) and while the effects of cat predation were well documented up to the early 1990s (Schramm 1983; Fugler *et al*. 1987; Van Rensburg and Bester 1988; Newton and Fugler 1989; Cooper and Fourie 1991; Cooper *et al*. 1995), recent estimates of petrel breeding success are lacking.

Since 2015, the dramatic increase in mouse predation on albatrosses at Marion Island has been of particular concern (Chapter 6). The hundreds of thousands of petrels that breed in burrows and lava caves are also likely to be attacked by mice, yet to date there has been little direct evidence of mouse predation on burrowing petrel chicks, probably at least in part because attacks on petrels nesting in underground burrows are much harder to detect that those on albatross chicks. Fugler *et al*. (1987) suspected that mice predated 'some eggs and small chicks' of blue petrels *Halobaena caerulea* at Long Ridge in 1982 when they found 'one chick carcass had deep wounds on the back of the neck, probably made by a mouse' (p. 106). On Gough Island mice have been shown to be very efficient predators of burrow-nesting petrel chicks (Wanless *et al*. 2012; Chapter 4), and there is circumstantial evidence that mice impact breeding success and distribution of storm petrels on Steeple Jason Island (Bolton *et al*. 2014), but the extent of mouse predation on burrow-nesting petrels on Marion Island is unknown. In this chapter, I report the breeding success of four species of burrowing petrels over one to five breeding seasons and assess the extent of mouse predation using video surveillance inside burrow chambers. Reasons for nest failures are summarized with a particular focus on the frequency of chick mortalities in the first 1–2 weeks after hatching. I hypothesize that (1) mice are suppressing the post-cat recovery of petrel populations through depredation of petrel eggs and chicks, and (2) petrel species that breed in winter are more severely affected by mouse predation than summer breeders, because mice face a greater challenge to obtain food in winter than in summer, similar to the pattern observed on Gough Island.

Methods

Fieldwork was conducted from April 2012 to March 2017 at Marion Island (293 km^2), south-west Indian Ocean. Four species of burrowing petrels were monitored for one to five seasons: blue petrels (one season), white-chinned petrels *Procellaria aequinoctialis* (two seasons), grey petrels *P. cinerea* (five seasons) and great-winged petrels (five seasons). Study nests were individually marked with numbered PVC poles and regular nest checks made with a burrow-scope (custom made burrow-scope with a high-resolution conical pinhole camera, light-emitting diode (LED) torch (200 lumens) and an 18×21 cm colour monitor) to record breeding success. The bright torch allowed sufficient image quality to monitor chicks for mouse wounds. Infrared video cameras (details below) were installed at a subsample of burrows to record activity inside the nest chambers. Access to nest chambers was facilitated by digging hatches over the entrance burrow ~ 0.3 m away from the nest chamber. These access hatches were cut to snugly accommodate a five litre plastic tub, which was filled with the vegetated 'plug' removed to cut the hatch. The tub with its live vegetation plug could then be removed and reinserted with minimal disturbance, crucially not revealing the location of the nest to brown skuas.

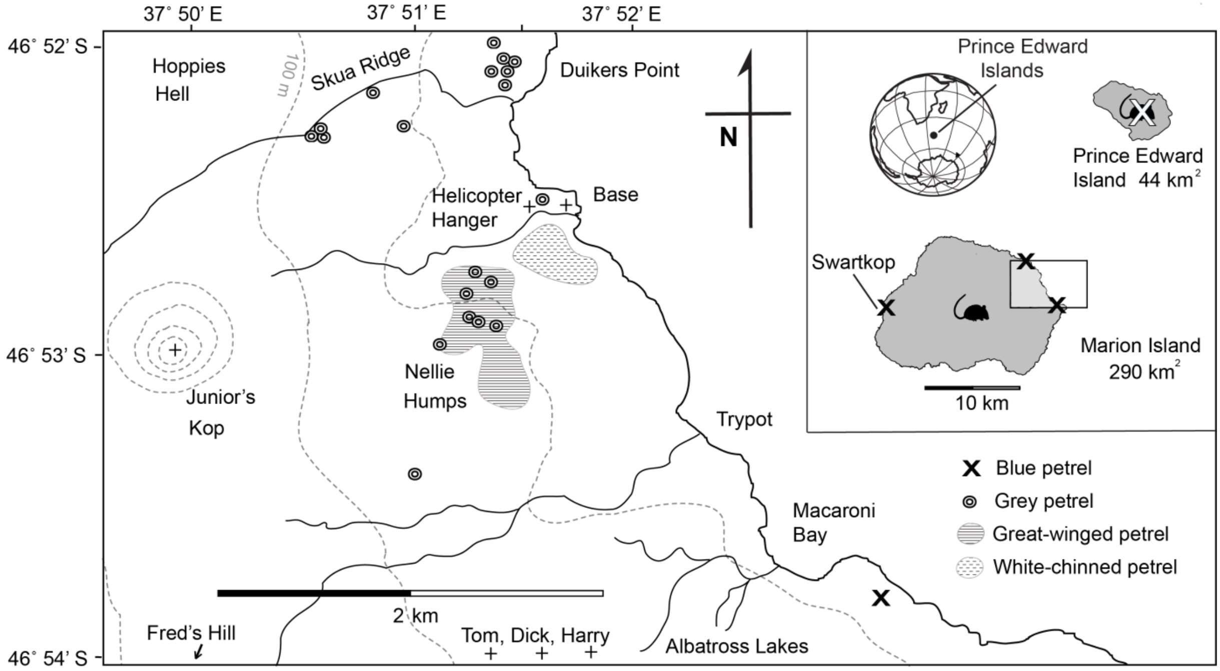

Figure 5.1. Study area in the north-east corner of Marion Island, showing the locations of the burrow-nesting petrel study areas. The insert shows the location of the Prince Edward Islands, with Prince Edward Island 22 km to the north-east of Marion Island.

Breeding success of blue petrels was estimated at three study sites (Fig. 5.1), which represent the main blue petrel breeding habitats (Appendix 1): 1) *Acaena* slopes at Macaroni Bay (46°53.432'S, 37°52.493'E), where the creeping stems of *Blechnum penna-marina* ferns and *Acaena magellanica* creepers form large soft mats of vegetation on well-drained soils; 2) *Leptinella* plains at Swartkop (46°55.380'S, 37°35.799'E), where there are extensive low herb fields of *Leptinella plumosa* and *Crassula moschata* with occasional large sprawling cushion plants *Azorella selago* on coastal slopes and flat areas with frequent sea spray; and 3) tussock slopes at Long Ridge (46°50.841'S, 37°49.098'E) dominated by tussock grass *Poa cookii*, tufts of the sedge *Uncinia compacta* and introduced grasses *Poa annua* and *Agrostis stolonifera*. At each site, 50 burrows containing incubating birds were individually marked and fitted with access hatches to view the nest chamber (see above). Study nests were selected at the end of September 2012 when birds had already started laying, thus early egg failures and accurate laying dates were not recorded. Study nests were monitored for one breeding season (2012/13, *n* = 150 breeding attempts) from early–mid-incubation until chicks fledged. At Macaroni Bay, nests were checked every 2 days from mid-incubation until chicks were 3 weeks old and weekly thereafter. At Long Ridge, nests were checked every 2 weeks from early incubation, but weekly at hatching. At Swartkop, nests were checked every 3 weeks from early incubation.

White-chinned petrels study burrows (Fig. 5.1) were located on coastal slopes dominated by *Blechnum penna-marina* ferns and patches of *Acaena magellanica* near the station (Base) and inland of Gentoo Lake (46°52.649'S, 37°51.572'E), and burrows for monitoring with nest cameras were located down-slope from the helicopter hanger (46°52.523'S, 37°51.436'E). Freshly renovated burrows were selected for the study and access hatches were fitted to 50 burrows in late October 2012, prior to laying when birds were on their pre-laying exodus. Eggs were laid in 37 of these burrows, so a further 13 burrows were selected after laying to make up 50 study burrows. Study burrows were monitored every 7–10 days from laying until chicks fledged over two breeding seasons (50 breeding attempts in 2012/13 and 41 in 2013/14), with more frequent checks (3–5 days) from hatching until chicks were 2 weeks old.

Grey petrels are scarce on Marion Island, where they nest singly or in small groups in burrows or in well concealed caves (Fig. 5.1). Most breeding caves are among large grey lava boulders (e.g. inland from Duikers Point, 46°52.041'S, 37°51.397'E), but nests were also found in black lava caves. Extensive searches of all possible burrows and caves found 20 nest sites (11 in caves, nine in burrows) within an ~ 300 ha area around the station (Fig. 5.1) in the early winters (April–May) of 2012–16. Useful clues to an active nest site were feathers lying near the entrance and fresh faecal stripes, often on a small steep slope covered by *Blechnum* where birds display at night. Grey petrels were responsive to call backs which were used to identify the occupants of suspected active looking burrows (see Chapter 5). Study burrows were monitored every 7–10 days from laying until chicks fledged over five breeding seasons (57 breeding attempts, 11 ± 2 (standard deviation, SD) per year), with more frequent checks (1–5 days) from hatching until chicks were 2 weeks old.

Great-winged petrels study burrows were located along the inland slopes at Nellie Humps (46°52.934'S, 37°51.365'E, Fig. 5.1), an area of undulating hummocks with well-drained soils dominated by *Blechnum penna-marina* ferns. An additional five burrows were selected down-slope from the helicopter hanger for monitoring with nest cameras. In 2012, 15 recently renovated burrows where fitted with an access hatch and checked every 2–5 days from 20 May to 20 June to monitor laying dates. Eggs were laid in nine of these burrows, with a further 48 occupied burrows selected after laying to make 57 study burrows. Study burrows were monitored weekly from laying until chicks fledged over five breeding seasons (276 breeding attempts, 55 ± 2 per year), with more frequent checks (every 2–5 days) over the laying, hatching and small chick stages.

Table 5.1. The number of burrow chambers filmed using permanent and mobile infra-red burrowcams for four species of burrow-nesting petrels at Marion Island from 2012–17.

	Blue petrel	**White-chinned petrel**	**Grey petrel**	**Great-winged petrel**
Breeding seasons monitored	1	4	4	5
Nests filmed	2	16	18	19
Complete breeding cycles filmed	2	16	2	12
Total days filmed	170	2108	492	1254
Average film days per nest ± SD	85 ± 9	132 ± 38	27 ± 66	66 ± 57
Chicks killed by mice	0	0	3	1
Chick failure rate due to mice	0%	0%	17%	5%

Filming nests with video surveillance cameras

Twelve small infrared cameras linked to digital video recorders were customized to film activity inside nest chambers. Each camera (B/W low light mini camera, code E-25B-B36, 1/3'' CCD) had a 2.1 mm wide angle board lens, covering 120°, accompanied by a ring of 12 infrared LEDs. Inspection hatches were dug through the roof of the burrow passage to gain access to the nest chamber. Each camera was housed in 40 mm PVC piping to keep it dry and to prevent mouse damage, secured to a metal angle-iron pole and positioned 20–30 cm away from the incubating bird.

Eight of these cameras were deployed in burrows on coastal *Blechnum* slopes within 200 m of the helicopter hanger, which allowed the cameras to be linked to the station by video cables. These cameras were motion activated and connected to a video surveillance system (SuperDVR software) which enabled a live feed, with footage recorded onto a computer. These long-term burrowcams were used to monitor complete breeding cycles and were installed in active white-chinned petrel burrows (16 breeding cycles filmed over the five year study period) in summer and moved to active great-winged (12) and grey petrel (two) burrows in winter (see Table 5.1 for details). Cameras were either installed into the burrow chamber before laying or at mid-incubation, when the disturbance of installation was less likely to cause the occupants to abandon their nest. Camera installation took <10 minutes and did not result in any immediate nest failures. Since I suspected the mice would depredate newly hatched and newly independent chicks, it was important to have the cameras *in situ* before hatching. The remaining four cameras were moved among burrows and sites to monitor small chicks <2 weeks old (the time when chicks are most vulnerable to mouse predation; see Chapter 4). Each motion activated camera was connected to an independent MemoCam (Video Domain Technologies; powered by 50 Ah 12 v battery; charged manually/solar; data storage micro SD). These mobile burrowcams were used to monitor inside 16 grey, seven great-winged and two blue petrel nest chambers over the study period (Table 5.1).

Data analysis

For all species, hatching success was calculated as the proportion of eggs that produced live chicks; this was a maximum estimate as not all eggs were monitored from laying. To account for this, I calculated daily rates of egg survival for each species over each season using the nest survival model in MARK (version 8.x; White and Burnham 1999). I estimated the corrected hatching success as the daily egg survival raised to the power of the length of the incubation period (Rotella 2009). Blue petrel incubation length (49.0 ± 2.0 days, *n* = 7) was taken from Fugler *et al*. (1987); I collected data on incubation lengths for white-chinned petrels (59.5 ± 1.9, *n* = 6), grey petrels (56.6 ± 1.5 days, *n* = 3) and great-winged petrels (55.6 ± 4.2 days, *n* = 6) in 2012/13. This method of estimating egg survival assumes that daily nest survival is similar across the incubation period within a study site (Mayfield 1975). Since all nests were followed from egg stages, fledging success was calculated as the proportion of hatched chicks that survived to fledge. The overall breeding success was calculated as the product of the estimates of hatching success and fledging success. Skuas predated 12 burrows by digging out the inspection hatches (2% of breeding attempts at burrows with inspection hatches over the study period: two white-chinned and ten great-winged petrel burrows). The installation of access hatches might have increased the risk of skuas digging up these burrows, thus these breeding attempts were excluded from analyses.

The video files recorded a date and time stamp which enabled us to record a detailed sequence of activity for each filmed nest, including hatching date, frequency of mouse visits/attacks, age of the chick when it was first left alone and the date/time of death for chicks that died before fledging. Video footage from the 2012–13 seasons was manually reviewed to calculate the visitation rate of mice in burrows with chicks 1–14 days old. To quantify the visitation rate, I counted each time a mouse entered the frame as a single mouse visit. This doubtless resulted in multiple records of the same mouse, but it provided an objective criterion to quantify visitation rates. When multiple mice were in the burrow at one time, each mouse counted as a separate visit. I analysed all the footage, thereby eliminating individual observer effects. Consequently, the method provided an index of visitation rates that could be compared between seasons and species.

Seasonal and inter-species differences between frequency of mouse visits to burrows in 2012 were tested using Kruskal–Wallis tests with $P < 0.05$ as the cut-off for significance. Means are presented ± SD unless stated otherwise. Breeding years refer to seasons (i.e. 2012 for the 2012–13 summer breeding season).

Results

Breeding success

Breeding success of blue petrels in the three study colonies in 2012 was 61 ± 6% (Tables 5.2 and 5.3), more than double the breeding success in the 1980s (Fig. 5.2). No direct evidence of mouse predation (chicks with mouse wounds) was found; however, 20–44% of failed eggs had mouse incisor marks on freshly broken egg shells, and small chick carcasses were scavenged by mice at all three sites (Table 5.3). Predation by skuas accounted for 50% (5/10) of the chick mortalities at Macaroni Bay, where the loose soil and low woody *Acaena* shrub provided little defence against burrow excavation by skuas. At Swartkop (*Leptinella*) and Long Ridge (*Poa*) there was a similar skua presence to the Macaroni Bay colony, but the proportion of failures due to skuas was lower (20% of chick failures at both sites), possibly due to the compact soil and dense summer growth of *Poa* and *Leptinella* vegetation which seemed to provide better protection to skua predation attempts.

Table 5.2. Breeding attempts (number per year (mean ± SD)) and overall breeding success (%; mean ± SD (range)) for four species of burrow-nesting petrels monitored from one to five seasons at Marion.

Species	Seasons	Breeding attempts	Breeding success
Blue petrel	summer 2012	150	61 ± 6% (54-66%)*
White-chinned petrel	summers 2012, 2013	91 (50, 41)	59 ± 6% (63%, 55%)
Grey petrel	winters 2012-2016	57 (11 ± 2)	33 ± 21% (0-56%)
Great-winged petrel	winters 2012-2016	276 (55 ± 2)	52 ± 7% (41-62%)

*at three locations in 2012

Breeding success of white-chinned petrels was 63% in 2012 and 55% in 2013 (Tables 5.2 and 5.3). No direct evidence of mouse predation was found during nest checks or recorded in the camera-monitored burrows, but in 2013, small chicks were found dead with mouse wounds on two occasions. Breeding success of grey petrels averaged 34 ± 21% (range 0–56%, *n* = 57 monitored breeding attempts) over the five breeding seasons (Tables 5.2 and 5.3). Most chick mortalities occurred in the first week after hatching and in the last three study years all were of very small chicks and were almost certainly due to mouse predation. At Duikers Caves in 2012, two large chicks died when almost fully grown with emergent flight feathers. One of the dead chicks was too deep in a narrow cave to retrieve, but burrow camera footage revealed no visible wounds or obvious mouse activity around the carcass. The other chick was retrieved and a post-mortem revealed no mouse predation wounds, very little body fat and an empty stomach. I suspect that these chicks died of starvation. Breeding success of great-winged petrels averaged 52 ± 7% (range 41–62%, *n* = 276 monitored breeding attempts) over the five breeding seasons (Tables 5.2 and 5.3). Chick mortality was highest in the first week after hatching and small chicks were found dead with mouse wounds in all five seasons.

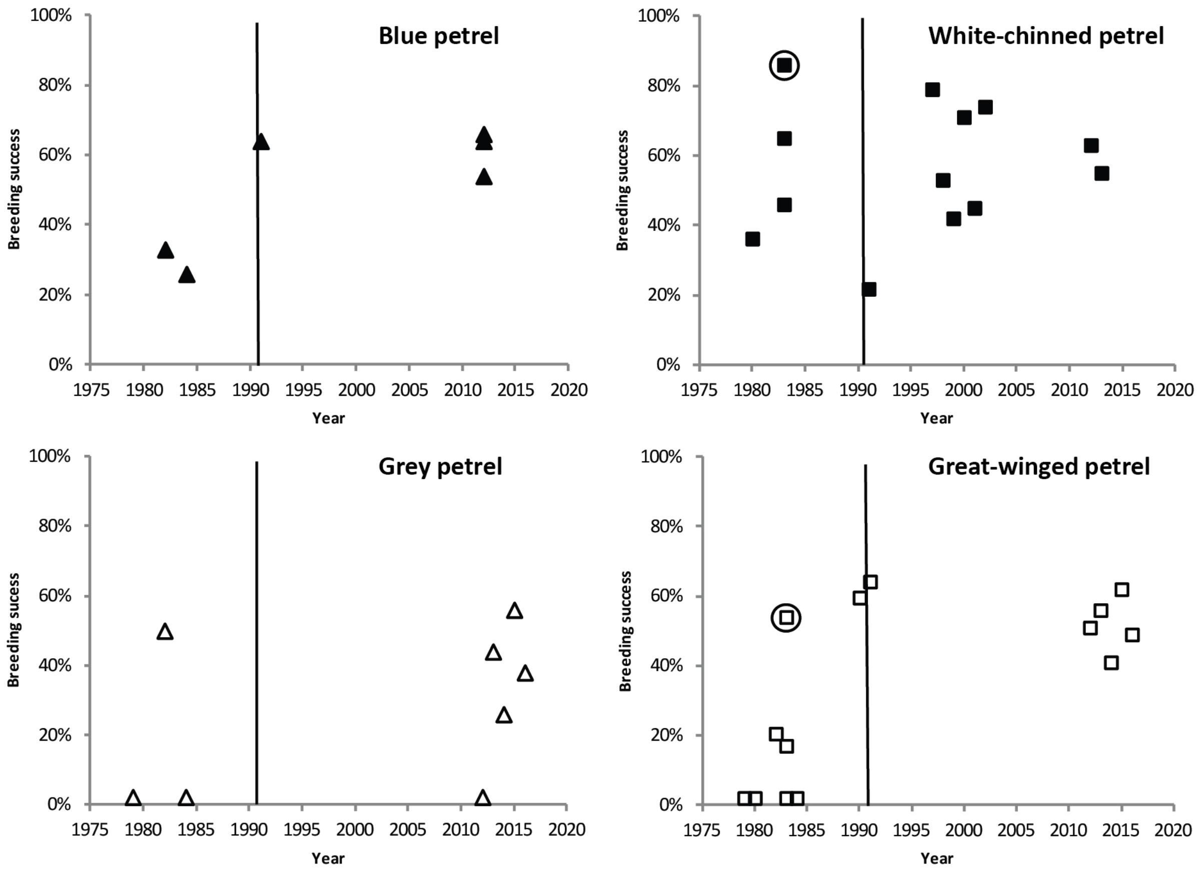

Figure 5.2. The breeding success of four species of burrow-nesting petrels at Marion Island from 1979–2016. Cat eradication efforts started in the 1970s and progressed through multiple phases until all cats were removed from the island in 1991 (vertical line). Circled data points indicate study areas in cat-free enclosures (Van Rensburg and Bester 1988; other data sources: Schramm 1983; Fugler *et al*. 1987; Newton and Fugler 1989; Cooper and Fourie 1991; Cooper *et al*. 1995; FitzPatrick Inst. unpublished data).

Mouse predation, frequency of mouse visits and temporary egg neglect

Mouse activity in the nest chambers of two blue petrel burrows were video recorded from early- to mid-incubation until both chicks fledged. Blue petrel chicks rarely reacted to a mouse entering the nest chamber and only very occasionally did a mouse make brief contact with the chick. Mice appeared to scavenge around the nest bowl. Egg neglect was recorded in one of the filmed burrows when the parent left its egg unattended for 49 hours (egg age approximately 32 days). Mice did not visit the burrow during this time and the chick hatched 15 days later. Egg neglect was also recorded for two nests in the Macaroni Bay study colony in early incubation (8–10 days after laying) when, over a sequence of nest checks every other day, the eggs were recorded as being incubated, to being left unattended and cold to the touch, to being incubated again. Both eggs hatched successfully. It is possible that other eggs were also temporarily neglected, but were eaten by mice before the adult returned (see Table 5.3 ‘Eaten with mouse teeth marks in shell’).

Table 5.3. The breeding success and probable causes of egg and chick mortality for four species of burrow-nesting petrels monitored at Marion Island from 2012–17.

Study species	**Blue petrels**			**White-chinned petrels**		**Grey petrels**					**Great-winged petrels**				
Study years	**2012**			**2012**	**2013**	**2012**	**2013**	**2014**	**2015**	**2016**	**2012**	**2013**	**2014**	**2015**	**2016**
Number of monitored burrows (egg)	50[a]	50[b]	50[c]	50	41	13	13	8	11	12	57	53	54	58	54
EGG MORTALITY															
Total egg mortality	9	5	7	12	11	6	3	2	2	4	14	8	21	12	20
Disappeared/went missing	2	0	2	1	0	0	0	0	0	0	0	0	1	2	1
Abandoned early in incubation	1	2	1	6	7	4	2	2	2	2	3	3	7	2	10
Abandoned after extended incubation	1	1	1	1	2	2	0	0	0	0	9	4	10	4	3
Eaten with mouse teeth marks in shell	4	1	2	0	0	0	1	0	0	2	2	0	0	1	3
Flooded nest chamber	0	0	1	1	1	0	0	0	0	0	0	0	1	0	0
Egg rolled off nest mound into mud	0	0	0	3	1	0	0	0	0	0	0	1	2	3	3
Burrow excavated by skua	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Egg exposure days	1349	1539	1927	2510	1718	378	281	305	348	331	2210	1859	1824	2285	2111
Hatching success (%)	72	85	84	75	68	40	55	69	72	50	70	76	53	75	59
95% confidence interval	69-75	82-89	81-86	73-78	64-73	30-54	46-65	58-82	69-76	43-59	66-73	74-79	48-57	72-77	55-62
CHICK MORTALITY															
Chicks hatched	41	45	43	38	30	7	10	6	9	8	43	45	33	46	34
Total chick mortality	10	10	10	6	6	7	2	3	2	2	12	13	7	8	6
Chick mortality first 7-14 days	1	4	4	3	4	3	1	3	2	2	5	7	5	4	5
Large carcass found mouse scavenged	3	1	2	0	0	1	1	0	0	0	2	0	2	2	1
Burrow excavated by skua	5	2	2	1	0	0	0	0	0	0	1	3	0	1	0
Chick killed by grey petrel	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Chick killed by white-chinned petrel	0	0	0	0	0	1	0	0	0	0	4	3	0	1	0
Large chick abandoned (starvation)	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0
Live chick found with mouse wounds	0	0	0	0	0	0	0	0	1	0	0	0	0	2	0
Unknown/chick disappeared	1	3	2	1	2	0	0	0	0	0	0	0	0	0	0
Chicks fledged	31	35	33	32	24	0	8	3	7	6	31	32	26	38	28
Fledge success (%)	76	78	77	84	80	0	80	50	78	75	72	73	79	82	82
Breeding success (%)	54	66	64	63	55	0	44	34	56	38	51	56	41	62	49
Mean breeding success (study period)	61 ± 6%			59 ± 6%		33 ± 21%					52 ± 7%				

Data are presented as *n*, unless otherwise stated.

Blue petrels were monitored for one season only (2012) at three different locations: Macaroni Bay (*Acaena*)[a]; Long Ridge (*Poa*)[b] and Swartkop (*Leptinella*)[c]

Sixteen white-chinned petrel nesting attempts were monitored with burrow cameras from 2012–17 (Table 5.1). None of the chick mortalities were due to mouse predation. Mice were observed in all filmed nest chambers relatively infrequently (compared to winter breeding species, Fig. 5.3) and the only direct contacts observed were of mice licking the chicks' down, presumably to glean spilt oil and food after a chick was fed. Chicks appeared to be fairly tolerant of this intrusion. They would occasionally sit up and bill snap; however, no defensive vomiting was recorded.

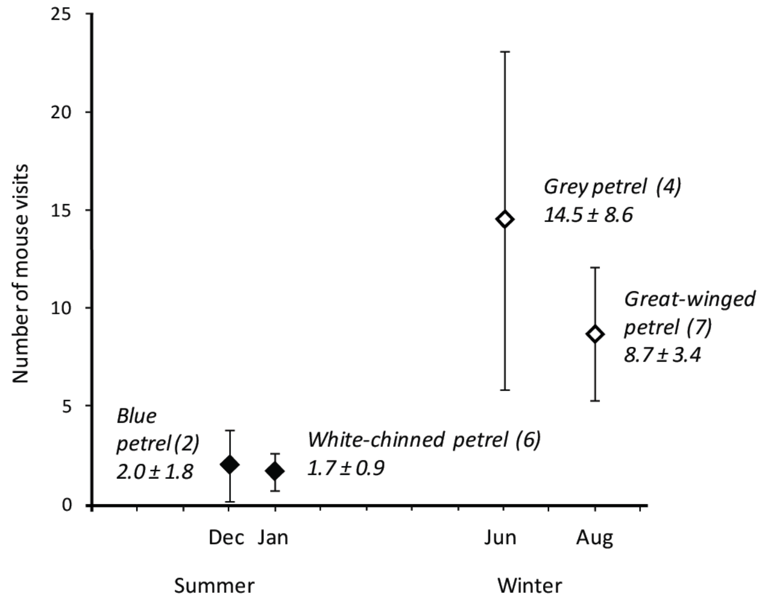

Figure 5.3. Relationship between the average daily visitation rates of mice in burrowing petrel study burrows during the first week after chicks hatched in the summer/winter of 2012–13 at Marion Island. Data from burrows monitored with infrared video cameras (see methods for details and visitation analyses); numbers in parentheses indicate number of monitored nests per species. Black diamonds indicate summer breeders and white diamonds indicate winter breeders. Data represent mean ± standard error.

Two complete grey petrel breeding cycles were recorded in a camera-monitored burrow near the helicopter hanger, with no mouse predation recorded in either year. Video footage showed that incubating birds often left their burrows for a short period (<10 minutes), usually in the early evening and more frequently in the week after hatching. In 2012 a female abandoned her newly laid egg in a cave at Nellie Humps and the egg was eaten by mice before the male arrived 2 days later. An additional 16 grey petrel nests were filmed from hatching for 1–41 days; three of these chicks were attacked and killed by mice (Table 5.1). Mouse visitation rates in 2012 when chicks were <7 days old were the highest recorded (Fig. 5.3; 14.5 per day, range 1–74 per day, *n* = 4 nests which had unbroken footage of the first week after hatching, no significant difference in visitation rates between grey petrel burrows, Kruskal–Wallis, $H_{4,\ 15} = 8.85$, $P = 0.904$). At three of the nests with the highest visitation rates, mice harassed the small chicks to such an extent that the chicks vomited oil and repeatedly shuffled around on their nest mound. Two of these chicks were dead the following morning and it is very likely that mice were the cause, yet conclusive evidence was not recorded due to camera failure (the camera wires had been chewed by mice). The other chick survived the first two weeks unwounded, but mice continued to frequently visit the burrow, especially when the chick was being fed by a parent (Fig. 5.4). This chick died at 12 weeks old and the freshly mouse scavenged carcass was found on the nest, but the cause of death was not confirmed. However, in 2015 video footage was obtained of a small chick (age <5 days) being attacked and killed while still being brood-guarded by its parent in Duikers Cave. This was the first conclusive video evidence of mouse predation on burrowing petrel chicks at Marion Island (see https://youtu.be/Og1d6a2cmXQ).

Figure 5.4. Images from infrared video footage of a grey petrel nest in a cave showing mice apparently licking oils spilt during parental feeds off the chicks' downy feathers (a) even in the presence of the parent (b) (photos Ben Dilley).

No mouse predation was recorded during 12 complete monitored breeding attempts of great-winged petrels in burrows near the helicopter hanger (Table 5.1). Incubating adults often left their burrows for short periods, especially shortly after laying, and in one case a mouse attempted unsuccessfully to eat a neglected egg. Mouse visitation rates at burrows with chicks <7 days old were high (Fig. 5.3; 8.7 ± 3.4 (SE), range 1–39 per day, *n* = 7 nests) compared to summer-breeding species (average <2 visits per day, range 0–9), but lower than visits to grey petrel burrows earlier in winter when mouse densities are higher (see discussion). In 2012, video recordings from mobile burrowcams showed mice aggressively and repeatedly harassing small chicks on four occasions, causing the chicks to shuffle around on their nest mounds to face the intruding mice while bill snapping and sitting upright. All four chicks survived and on closer inspection none had mouse injuries. On 20 July 2015, a small newly independent chick (<5 days old) was filmed being attacked by two mice in a burrow at Nellie Humps (Fig. 5.5). The chick was dead within 24 hours of being attacked (see https://youtu.be/D9vPoFsjvgs).

Figure 5.5. Two-day-old great-winged petrel chick attacked and killed by two mice within hours of being left alone by its parent after the brood-guard phase on 20 July 2015 at Nellie Humps, Marion Island (photo Stefan Schoombie).

In summary, winter breeders had lower breeding success than summer breeders (Table 5.2), with most chick fatalities of small chicks <14 days old (Table 5.3). Mice were filmed attacking and killing chicks of both winter-breeding species: grey (3/18 nests filmed; 17%) and great-winged petrels (1/19; 5%). These are the first confirmed records on video of fatal mouse attacks on burrow-nesting petrel chicks at Marion Island. Mouse predation was suspected previously, when small chicks were found dead with fresh wounds typical of those inflicted by mice (open wounds mainly to the back, rump or head; Chapter 4). Winter breeders were worse affected by mouse predation than summer breeders, and this was related to higher mouse visitation rates to petrel burrows in winter. In 2012, mouse visitation rates to burrow chambers containing chicks <7 days old were significantly higher (Kruskal–Wallis, $H_{2,\ 29}$ = 67.34, *P* < 0.001; Fig. 5.3) for winter breeders (10.9 ± 12.8, 1–74 visits per day, *n* = 11 burrows) than summer breeders (1.8 ± 2.5, 0–9, *n* = 8).

Discussion

Predation on petrel chicks

While there is mounting evidence of an increase in mouse attacks on surface-nesting albatross chicks at Marion since the early 2000s (Jones and Ryan 2010; Chapter 6), few direct records of mouse interactions with burrow-nesting petrels existed because of the technical difficulties of observing inside burrows. This study illustrates how mouse predation impacts the breeding success of burrow-nesting petrels at Marion Island. As expected, winter-breeding petrels were affected to a greater extent than those species that breed in summer. The magnitude of the impact on the breeding success of at least grey petrels probably is sufficient to limit population growth and explains why grey petrels show no evidence of a population recovery since cats were eradicated from Marion Island in 1991 (Chapter 3).

Small chicks of winter-breeding grey and great-winged petrels were filmed being attacked and killed by mice. Chicks were dead within hours of being attacked and carcasses were usually consumed completely, leaving little evidence as to the reason for the nest failure. This could explain why so few chicks injured by mice have been found during routine nest checks relative to the high proportion of small chick fatalities, whereby many chicks ‘disappear’ between nest checks. Summer-breeding white-chinned and blue petrels appear to be less affected, with few small chick fatalities in the first weeks after hatching (Table 5.3) and lower mouse visitation rates inside burrows compared to winter-breeding species. Summer conditions provide more abundant food supply (e.g. invertebrates, vegetation) for mice, and as such mice are less desperate for food, but as winter sets in mice have limited food resources (Cuthbert *et al*. 2016) and have resorted to eating seabird chicks (Chapter 4; Appendix 3). A similar process probably occurs at Marion Island. In 1992, Avenant and Smith (2003) estimated the per capita food availability for mice (macro-invertebrate biomass per mouse) to be ~ 3.4 kg·ha^{-1} in biotic habitats (mostly coastal areas where the vegetation is heavily influenced by seals and seabirds; Gremmen and Smith 2008) and 3.6·kg ha^{-1} in mire habitats (boggy areas, ranging from wet to dry mires; Gremmen and Smith 2008) in early summer, but in early winter the per capita food availability was <10% of the summer estimates (0.4 kg·ha^{-1} and 0.2·kg ha^{-1}, respectively). Grey petrel chicks hatching in early winter had the highest level of mortality of small chicks, at a time when mouse densities are still fairly high but food availability is low, resulting in the lowest seasonal food availability per capita for mice. In the last three study years, all of the grey petrel chicks that died were <7 days old and this is compelling evidence that mice are contributing to these mortalities. Great-winged petrel chicks hatch 1–2 months later than grey petrels, when mouse numbers have already fallen, explaining the better breeding performance of great-winged petrels.

There are no estimates of adult petrel survival rates on Marion, but their breeding success increased immediately following the removal of cats (Fig. 5.2) and remains at moderate levels which suggests that petrel populations have the potential to recover. However, the recent repeat survey of burrow densities showed only a modest recovery of most burrow-nesting petrel populations since cats were eradicated 25 years ago, with no evidence of an increase in grey petrels (Chapter 3). Grey petrels are drowned accidentally on long-lines, which might also contribute to their failure to recover after cats were removed from Marion Island. However, the closely related white-chinned petrel is killed in much larger numbers by fisheries in the region (e.g. Petersen *et al*. 2009), and yet its population has shown the fastest growth following the removal of cats (Chapter 3). Across all petrels, the changes in burrow density (Chapter 3) and breeding success results in this study show a similar pattern; summer breeders have higher breeding success and recover faster than winter breeders, suggesting there is a common factor suppressing the recovery of winter-breeding petrels.

The breeding success estimates for blue and white-chinned petrels are within the ranges reported elsewhere. I did not find any live blue petrel chicks with mouse injuries, but most of the chick mortalities I recorded were very small chicks, similar to those I found on Gough Island, where mice are significant predators of petrel chicks (Wanless *et al*. 2012; Chapter 4). The low breeding success of blue petrels in the early 1980s (Fig. 5.2) was largely caused by cat predation because at that stage the cat control programme was in its early phases and it was estimated that there was a 70% increase in cat predation on blue petrels from 1975–1982 (Van Rensburg 1985). Once cats were eradicated, blue petrel breeding success improved (Fig. 5.2) and although too few data exist since 1991 to show any long-term trend, the levels of breeding success on Marion in 2012–13 appear to be within or even above the range reported for blue petrels at Mayes Island, Kerguelen Archipelago, where mice are also the sole introduced mammal (Chastel *et al*. 1995). Chastel *et al*. (1995) reported that breeding success varied significantly from 1986–94 (26–65%) and hatching failure (52%) accounted for 80% of the total breeding failures, primarily due to egg desertion, especially in years when birds showed poor body condition at the start of the breeding season. From 2012–14, breeding success of white-chinned petrels was similar to that recorded from 1997–2002, following the cat eradication on Marion, when breeding success averaged 61% (42–79%, *n* = 26–53 study nests per year, Fig. 5.2). These values are within the range reported from other sub-Antarctic breeding sites: at Bird Island (free of introduced predators), South Georgia, breeding success varied from 12–54% at two different study sites (n = 72 and 40 burrows) in 1985 (Hall 1987) and was consistent from 1996–98 at 44% despite inter-annual variation in the availability/abundance of Antarctic krill *Euphausia superba* (Berrow and Croxall 1999); and at Ile de la Possession, Crozet archipelago, where black rats *Rattus rattus* are known to depredate seabird chicks, breeding success was 55–79% at sites where rats were poisoned and 30–61% at control sites (Jouventin *et al*. 2003).

Although grey petrels are locally common on nearby Prince Edward Island (Ryan and Bester 2008), they are scarce on Marion with a very low nest density. Grey petrel chicks are particularly vulnerable to mouse predation since they hatch in early winter when mice have few alternative food sources (McClelland *et al*. 2017). Grey petrels were considered 'not common' on Marion at the start of the cat era in 1952 (Rand 1954), but their numbers were depleted over four decades of cat predation. It is unclear why the two large chicks were abandoned at Duikers Caves in 2012; it is possible that the adults were killed at sea. One banded pair has not been re-sighted and the nest site has since been used by a different pair. The average breeding success over this study period (34 ± 21%) was similar to grey petrels on Gough Island (34%, Chapter 4), where mice also prey on chicks. This suggests that mice are a major source of breeding failure for this species on Marion Island, which has shown the least evidence of recovering since cats were eradicated (Chapter 3).

The breeding success of great-winged petrels improved dramatically following the eradication of cats (Cooper and Fourie 1991) and remains at moderate levels (Fig. 5.2). However, like grey petrels, most breeding failures occur as a result of chick mortality in the first week after hatching (Table 5.3), likely to be principally due to mouse predation. On rodent-free Whale Island, New Zealand, closely related grey-faced petrels *Pterodroma [macroptera] gouldi* achieved 65% breeding success in 2000 and the population has apparently more than doubled since Norway rats *Rattus norvegicus* and rabbits *Oryctolagus cuniculus* were eradicated in 1985–87 (Imber *et al*. 2003b).

Temporary egg desertion

Temporary egg desertion has been documented for many burrow-nesting *Procellariiforms*, and eggs may still hatch despite being neglected for up to two days (Ancel *et al*. 1998). However, mice have been recorded to eat unattended eggs within hours (Campos and Granadeiro 1999; Chapter 4). The two temporary abandonments of blue petrel eggs recorded 8–10 days after laying were probably females unable to incubate any longer until relieved by their partners. The reason for the 49 hour egg abandonment in another nest only two weeks before it hatched is less clear, but blue petrels are known to leave their egg unattended temporarily throughout the incubation cycle (Ancel *et al*. 1998). Although in these cases the unattended eggs were not attacked by mice, I know this occurs from evidence of incisor marks in broken shells. It is unclear how large an egg that mice can successfully gnaw into. On Gough Island, mice are able to gnaw into the eggs of great shearwaters *Ardenna gravis*, which average 80×52 mm (pers. obs. 2013–2014), and probably grey petrels, which average 82×55 mm (pers. obs. 2013–2014). However, mice on Gough are larger than mice on Marion Island (Cuthbert *et al*. 2016), thus Marion mice might not be able to access such large eggs. Elsewhere, Imber (1976) reported that that Norway rats 'ate many abandoned eggs' (p. 58) that had been temporarily abandoned by grey-faced petrels for 1–6 days on Whale Island, New Zealand, and Campos and Granadeiro (1999) reported that mice ate almost half of the white-faced storm petrel *Pelagodroma marina* eggs on Selvagem Grande Island, with most being eaten within 24 hours of being left unattended.

Conclusions

Mouse biomass on Marion Island increased from 1990–2008 (Ferreira *et al*. 2006; McClelland *et al*. 2017), yet invertebrate biomass declined >80% since the late 1970s (McClelland *et al*. 2017), driven in part by a warmer, drier climate (Le Roux and McGeoch 2008) and also by the combined impacts of invasive species disrupting the ecosystem functioning (Chown and Smith 1993). Since 2015, there has been a definite increase in the frequency of mice utilizing surface-nesting seabird chicks as an additional food source (Chapter 6) and if invertebrate biomass continues to decline, the impact of mouse predation on Marion's seabird chicks may become increasingly significant.

I conclude that mice are suppressing the recovery of burrow-nesting petrel populations, especially those that breed in winter, through depredation of petrel eggs and chicks. The widespread increase in mouse predations on albatross chicks at Marion in 2015 is cause for concern and these results show burrow-nesting petrels are also at risk, providing further motive for the eradication of mice from Marion Island.

Chapter 6

‘Scalping’ of albatross fledglings by introduced mice spreads rapidly at sub-Antarctic Marion Island

Dead and injured grey-headed albatross *Thalassarche chrysostoma* chicks with mouse-inflicted ‘scalping’ wounds on Marion Island’s Grey-headed Albatross Ridge in 2015 (photo Ben Dilley).

This chapter is broadly based on this publication:

Dilley, B.J., Schoombie, S., Schoombie, J. and Ryan, P.G., 2016. ‘Scalping’of albatross fledglings by introduced mice spreads rapidly at Marion Island. *Antarctic Science, 28*(2), pp.73-80.

Author contributions: BJD & PGR planned the field work at the study site; All authors collected data in the field; BJD analysed the data and wrote the draft; PGR assisted with data analyses and manuscript edits/preparation. Subsequent data in 2016-2018 were collected by field workers on Marion Island.

Chapter 6: ‘Scalping’ of albatross fledglings by introduced mice spreads rapidly at sub-Antarctic Marion Island

Abstract

House mice *Mus musculus* were introduced to sub-Antarctic Marion Island more than two centuries ago, and have been the only introduced mammal on the island since 1991 when feral cats were eradicated. The first mouse injured wandering albatross *Diomedea exulans* chick was found in 2003 and since then attacks have continued at a low level affecting <1% of the population. In 2009 the first ‘scalpings’ were detected: sooty albatross *Phoebetria fusca* fledglings were found with raw wounds on the nape. In 2015 mice attacked large chicks of all three albatross species that fledge in autumn and these attacks have continued in each successive year since, but at varying intensities: grey-headed *Thalassarche chrysostoma* (at least 32–142 wounded chicks/year in 2015-2018; 1.1–6.2% of fledglings), sooty (15–50; 1.1–4.7%) and light-mantled *P. palpebrata* (1–4; 4.1–6.1%) albatross chicks. Filming at night confirmed that mice were responsible for wounds. Attacks started independently in small pockets all around the island’s 72 km coastline, separated by distances hundreds of times greater than mouse home ranges. The widespread nature of mouse attacks since 2015 on large, well-feathered chicks is alarming and highlights not only Marion Island as a priority island for mouse eradication but also that mice alone may significantly affect threatened seabird species.

Introduction

One of the major threats to oceanic seabird species is the introduction of mammalian predators such as rats *Rattus* spp., cats *Felis catus* and house mice *Mus musculus* onto their breeding islands (Croxall *et al*. 2012, Chapter 1). Rodents have been introduced to many oceanic islands and the devastating effects of rats on small to medium-sized seabirds are well known (Atkinson 1985; Jones *et al*.2008). Larger seabirds such as albatrosses are less affected by rat predation, although rats have attacked Laysan albatrosses *Phoebastria immutabilis* on Kure Atoll (Courchamp *et al*. 2003) and also may affect Amsterdam Island albatross *Diomedea amsterdamensis* (Thiebot *et al*. 2014).

Mice are estimated to have invaded more oceanic islands than rats, yet until recently they were considered to have little impact on seabird populations with only a few records of mice killing storm-petrels (Campos and Granadeiro 1999; Ainley *et al*. 1990) and petrels (Fugler *et al*. 1987; Table 1.1). However, observations on Gough Island over the last decade show predation by mice on albatross chicks and on petrel chicks and eggs is widespread, highlighting how mice can be devastating predators of seabirds when they are the only introduced mammal (Cuthbert and Hilton 2004; Hilton and Cuthbert 2010; Wanless *et al*. 2007; 2012; Cuthbert *et al*. 2013a, b; Chapter 4; Appendix 3). In the absence of competition and predation from larger introduced species, mice attain very high population densities, and resort to attacking and killing seabird chicks mainly in winter when other food resources are scarce (Cuthbert *et al*. 2016).

Sub-Antarctic Marion Island is located in the southern Indian Ocean and is a globally important breeding site for albatrosses, supporting some 22% of the world population of wandering albatrosses, 7% of grey-headed albatrosses *Thalassarche chrysostoma*, 9% of sooty albatrosses *Phoebetria fusca* and 3% of light-mantled albatrosses *P. palpebrata* (Tickell 2000; Ryan *et al*. 2009a). Mice were brought to Marion Island during the sealing era sometime before 1818 and were the sole introduced mammal until 1949, when cats were introduced to control mice at the newly-established weather station (Cooper 2008). The cats soon turned feral, greatly reducing burrowing petrel populations over four decades (Schramm 1986), before finally being eradicated by 1991 (Bester *et al*. 2002). This left mice as the sole introduced mammal on Marion Island. The first signs of mouse attacks on seabirds were recorded in the winter of 2003, when wandering albatross *Diomedea exulans* chicks were observed with rump wounds typical of those inflicted by mice on Tristan albatross *D. dabbenena* chicks at Gough Island (Jones and Ryan 2010). In April 2009 I observed that one third of sooty albatross fledglings at an isolated colony in the southwest of Marion Island were found 'scalped' with raw, bleeding crowns and necks, and a similar wound was found on a sooty albatross chick on the island's southeast coast (Jones and Ryan 2010). Mice were suspected of being responsible for these wounds (Jones and Ryan 2010), even though summer-breeding albatross chicks are seldom attacked by mice on Gough Island (Cuthbert *et al*. 2013a). In April 2010, I found another sooty albatross fledgling with scalp wounds at the same colony where multiple scalpings occurred in 2009, but no further attacks were recorded until 2015.

In this chapter, I confirm that mice can cause fatal wounds on albatross chicks at Marion Island, and report the unprecedented increase in the frequency and distribution of mouse attacks on albatross chicks in the autumn of 2015. I also report the pattern of spread of attacks in subsequent years (2016–2018). Scalping allows mice to attack well-feathered albatross chicks, raising concerns about the conservation status of all albatrosses breeding on the island.

Study area and Methods

The populations of albatrosses breeding at Marion Island (293 km^2, 46°45´S 37°45´E) have been monitored since the early 1980s (Ryan *et al*. 2009a). Approximately 1,850 pairs of wandering albatrosses breed each year in loose colonies on the coastal plains around the island, 7,500 pairs of grey-headed albatrosses breed on cliffs along the south coast, and 1,800 pairs of sooty and up to 400 pairs of light-mantled albatrosses breed singly or in small colonies on cliffs around the island (Ryan *et al*. 2009a; FitzPatrick Inst. unpublished data). Two to five ornithological field researchers are based on the island year round and conduct complete island counts of incubating adults and of large chicks to estimate crude breeding success. More accurate estimates of breeding success are obtained from three study colonies of wandering albatrosses (~270 pairs per year, initiated in the 1980s), one study colony of grey-headed albatrosses (~100 pairs per year, initiated in 1997), and five study areas to monitor sooty albatrosses (~120 pairs of sooty and ~20 pairs of light-mantled albatrosses, initiated in 2013). Chicks in these colonies are visited every few weeks until fledging.

The over-wintering field researchers spend a considerable amount of time in the field outside of study colonies, and further observations outside of colonies were provided by field workers from other research programmes who were asked to report wounded albatross chicks. Despite this ongoing surveillance, there have been few observations of mouse wounded albatross chicks (Jones and Ryan 2010). However, during the April 2015 relief voyage I joined the field team for the pre-fledging count of grey-headed albatrosses and we observed one fledgling on Grey-headed Albatross Ridge with head wounds typical of those seen on sooty albatross chicks in 2009 (Jones and Ryan 2010). This triggered a series of additional surveys of all grey-headed and coastal sooty albatross colonies. All three small, summer-breeding albatross species breed on cliffs, limiting access to most colonies. Chicks were examined with binoculars for signs of mouse attacks. Most observations were made from cliff tops, but a few colonies were also inspected from below. Where possible, researchers in our field team entered colonies where attacked birds or carcasses were observed to check for cryptic wounds and to remove carcasses. In 2016–2018, similar checks were made each year, starting in February-March to determine when mouse attacks first started. Multiple checks through April and May determined how attacks spread around the island.

Grey-headed albatrosses

Detailed observations were made on grey-headed albatross chicks on Grey-headed Albatross Ridge, where most colonies are accessible on foot. Chicks in these colonies were checked for mouse wounds on 4–5 occasions from early March to late May, recording the number of wounded chicks and the nature of their wounds at each attack site. In order to gain a better understanding of the frequency of mice attacks, a sample of nests (17–30 in years 2015–2017) where chicks were seen with wounds were marked with poles to monitor the fate of the chicks. In years 2016 and 2017 wounded chicks were banded to monitor how many died and identify carcasses in subsequent checks. I confirmed the cause of these wounds by filming two wounded chicks with motion-activated infra-red cameras (Bushnell Trophy Trap Camera). Cameras were mounted 30 cm off the ground on PVC poles, 2–5 m from the nest, and set on high motion sensitivity to take one image per second for three seconds upon activation (following methods in Appendix 3). In addition, direct observations of the behaviour of wounded chicks at night were made in April-May in 2015 and 2016. Grey-headed albatross colonies on the slopes of Rook's Peninsula and Rook's Bay were checked on 4–5 occasions in each year, however in three of the years the last check was incomplete due to bad weather. The small grey-headed albatross colony in Crawford Bay was not checked because it could not be approached closely enough to assess whether any chicks were wounded.

S*ooty and light-mantled albatrosses*

Sooty albatrosses breeding along coastal cliffs are hard to count accurately because their dark plumage blends with the cliffs and nest sites are hard to access (Ryan *et al*. 2009a). Experienced observers in our field team worked systematically around the island's coast, counting and inspecting chicks. Where possible, observers descended into colonies for closer inspections and to remove carcasses. Complete surveys of coastal colonies were conducted on 2–4 occasions from late February to late May each study year, with additional checks (1–5) at some areas to follow the progress of wounded chicks. A remote camera was used to confirm that mice caused the head wound on one sooty albatross chick at Storm Petrel Bay on 18 May 2016. Light-mantled albatrosses mainly breed at scattered locations inland on Marion Island; chicks in only a few of these areas were checked for mouse wounds, but the small numbers of chicks on coastal cliffs were checked during surveys of sooty albatrosses.

Wandering albatrosses

In addition to regular checks of the three wandering albatross study colonies, a complete survey of all chicks from the Base Station to Cape Davis and from Mixed Pickle Cove to La Grange Kop was conducted from 25–30 June 2015, where each chick was inspected for wounds. Remote cameras were used to monitor chicks in 2012 (n = 12 chicks), 2013 (10), 2014 (6), 2015 (10), 2016 (6) and 2017 (1).

Attack sites

On each check of grey-headed and sooty albatross colonies, the locations and numbers of wounded chicks were noted and fresh carcasses were counted and removed. Carcasses were considered to be at an attack site if the carcass was within 20 m of a wounded chick (or beneath the site of known wounded chicks on sheer cliffs). Waypoints of attack sites were recorded using a Garmin GPS to assess the horizontal distance between attack sites, and these estimates are thus conservative especially for adjacent sites along steep slopes. Sites were considered discrete if the nearest adjacent attack site was >50 m away (the approximate home range of mice on Gough Island, Cuthbert *et al*. 2016).

Data analyses

In 2015, I assessed the proportion of carcasses found at colonies with mouse wounded chicks for grey-headed (83%) and sooty (89%) albatrosses in relation to the total carcass count. I did not re-assess this in subsequent years (2016-2018), but based on the 2015 findings considered all carcasses within 20 m of a known attack site to be linked to mouse predation. Also, in March-April 2016 numerous grey-headed albatross chicks were found dead from starvation (Vanstreels *et al.* 2018) which complicated re-assessing the proportion of carcasses linked to mouse predations.

To compare the overall impact of mouse attacks on albatross chicks (by species) between years I used the complete island counts of fledglings. For the analyses of spread, timing and frequency of mouse attacks and the number of affected sub-colonies, I used data collected over repeat island checks for sooty albatross, but for grey-headed albatross I used data from only Grey-headed Albatross Ridge as this was checked more frequently and search effort was comparable between years (repeat checks along the Rook's colonies were less complete and hindered by bad weather and thus represent the minimum numbers of affected chicks).

Results

Filming at night confirmed that mice were responsible for the wounds on all three albatross species where cameras were deployed (two grey-headed, one sooty and three wandering albatross chicks in 2015). At night, wounded grey-headed albatross chicks remained standing while other chicks slept (Fig. 6.1a), presumably to deter mice from attacking. Cameras recorded at least one mouse feeding on each wounded chick once they finally lay down. At Grey-headed Albatross Ridge, the two wounded grey-headed albatross chicks I filmed in May 2015 eventually gave up standing and sat at 18h10 and 03h00, whereupon mice climbed onto their heads. The chicks initially appeared to try to shake off the mice, but after a while the chicks sat quite still while the mice fed on their heads (Fig. 6.1b). A similar behaviour was observed on a grey-headed albatross chick early on the morning of 1 May 2016. The filmed sooty albatross chick was attacked from shortly after dusk; mice fed on the chick's head for four hours as it roosted with its bill tucked under its back feathers before the chick stood up, shook the mice free and remained standing for the rest of the night. This chick was still alive 6 days later. The three wounded wandering albatross chicks were all fed on by mice at night, although the chick filmed in July 2013 was killed the following day by giant petrels *Macronectes* sp. The chick with a head wound filmed in June 2015 was too young to stand for long periods; it was attacked by mice from shortly after dark, and died the day after it was filmed being attacked.

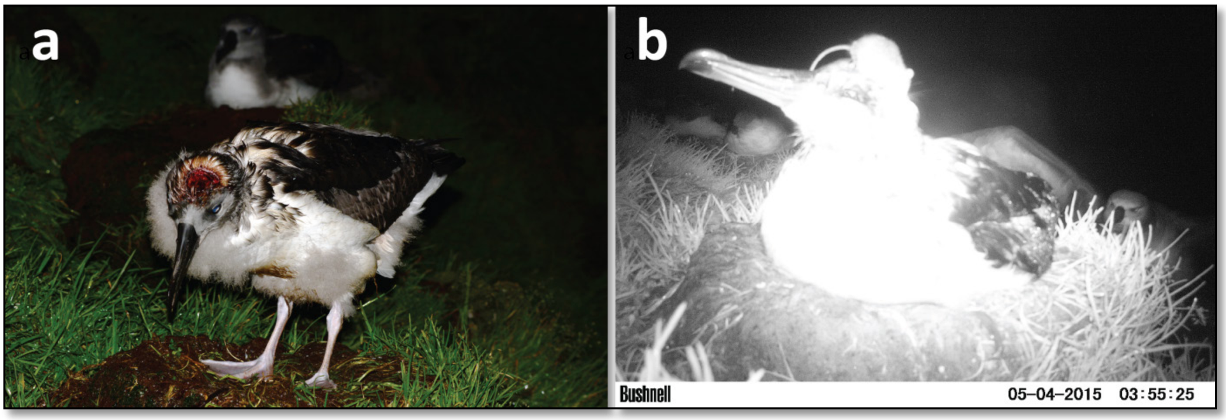

Figure 6.1. A wounded grey-headed albatross chick standing at night while an unwounded chick (in the background) lies down (a; photo Ben Dilley). Once the wounded chick sat down at 03h00 it was attacked by a mouse (b; photo Ben Dilley using infra-red remote-trigger). Both images are of the same chick and were taken on Grey-headed Ridge on the night of 3–4 May 2015.

The combination of their wounds and their high activity levels at night (standing) caused wounded grey-headed and sooty albatross chicks to appear weak and tired during the day compared to uninjured birds, which were lively and spent much time exercising their wings. When approached, badly wounded chicks failed to stand and bill-clap at the intruder, lying slumped over their nest with drooped wings. Wounded chicks also often had oily and dishevelled feathers, making them conspicuous even from a distance. However, birds with small wounds were easy to overlook, especially if wounded on the back of the head, because once alert they turned to face an intruder. The likelihood of detecting wounded chicks also varied among colonies, depending on how closely the colonies could be approached. Most colonies were scanned with binoculars from <50 m, but some were only possible to scan from greater distances (up to 200 m). As a result, the numbers of wounded birds reported here are minimum estimates.

Grey-headed albatrosses

The first wounded grey-headed albatross chick on Marion Island was observed during the annual fledgling census on 16 April 2015. Subsequent island-wide checks from 27 April to 26 May 2015 found 102 wounded chicks (4.6% of the island fledgling count; Table 6.1). Mice mainly targeted the head and neck: 63% of wounds on the crown, 22% on the nape, 9% on the back of the neck and 2% below the eye (n = 57 chicks, 11 with multiple wounds). The only attacks away from the head and neck were on the elbow joints (4% of chicks). Of the 17 chicks at marked nests, seven died before they were checked again 5.4 ± 2.0 days later (range 3–8 days); the other 10 chicks were still alive up to 11 days later. Most surviving chicks had enlarged wounds, although the rate at which wounds grew varied considerably. Some small crown wounds seemed to remain the same size at subsequent checks, whereas wounds on other chicks grew rapidly.

Table 6.1. Estimated numbers of grey-headed albatross chicks attacked by mice on Marion Island from 2015–2018. The percentage of chicks attacked is estimated as (minimum number of wounded chicks)/island chick count; percentage affected is estimated as (minimum number of wounded chicks + nearby carcasses at attack sites)/island chick count.

Count of grey-headed albatross	**2015**	**2016**	**2017**	**2018**
Island chick count	2201	2300	2291	2979
Minimum number of wounded chicks	102	142	33	32
Carcasses at or nearby attack sites	145	277	56	6
Carcasses away from attack sites	30*			
Sum of wounded chicks + nearby carcasses	247	419	89	38
% carcasses linked to mice	83%			
% of chicks attacked	4.6%	6.2%	1.4%	1.1%
% of chicks affected (wounded + carcasses)	11.2%	18.2%	3.9%	1.3%

* percentage carcasses linked to mice only assessed in 2015

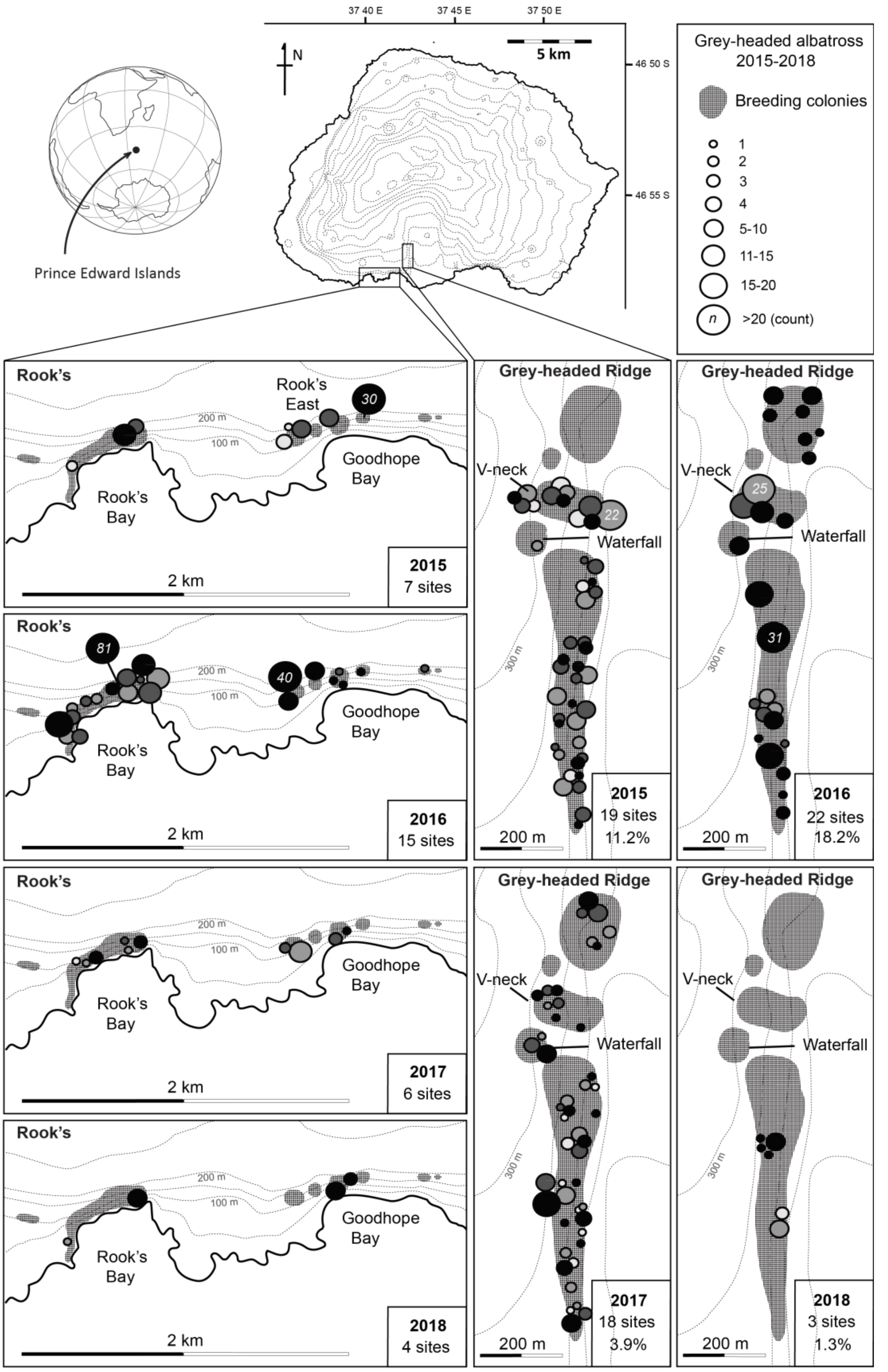

Figure 6.2. The locations of mouse attack sites at grey-headed albatross breeding colonies at Marion Island in 2015–2018. The shades inside the circles indicate when wounded chicks were counted in successive island surveys: first island survey (light grey), second (medium grey), third (dark grey) and final (black). Note the size of the circles at each attack site indicates the sum of the wounded chicks and carcasses found at or nearby affected sites. The percentage affected is the (minimum number of wounded chicks + carcasses at all attack sites)/island chick count.

Wounded chicks often occurred in clusters (Figs. 6.2 and 6.3). In 2015, 19 attack sites were located on Grey-headed Albatross Ridge (Fig. 6.2) containing 57 wounded chicks (5.5% of all chicks on the ridge, 3.2 ± 2.4 chicks per site, range 1–10 chicks per site). Attack sites farther inland had the greatest number of injured chicks. One of the uppermost attack sites had 10 wounded chicks within a ~30 m radius. Attack sites were 74 ± 30 m from the nearest adjacent attack site (range 51–150 m, *n* = 11). Seven attack sites were found in other grey-headed albatross colonies: three at Rook's Peninsula East, three at Rook's Peninsula West and one above Rook's Bay (Fig. 6.2).

In addition to wounded chicks, 175 grey-headed albatross chick carcasses were found, of which 145 (83%) were at known mouse attack sites. Taken together, the wounded chicks and carcasses suggest that mice attacked more than 11% of pre-fledging chicks in 2015, and most of these chicks died. The number of carcasses within attack sites along Grey-headed Albatross Ridge increased between checks, suggesting that the frequency of attacks increased as winter approached. On the final check on 23 May 2015, mice were frequently observed running within the colony during the day and four freshly dead chicks with mouse injuries were found on their nest mounds. These carcasses were untouched by other predator/scavengers such as brown skuas *Stercorarius antarcticus* or giant petrels.

Figure 6.3. Two grey-headed albatross chicks with typical 'scalping' crown wounds on Grey-headed Ridge, Marion Island, on 3 May 2015 (photo Ben Dilley).

In 2016, no wounded chicks were seen in February or March during the island checks. Subsequent island-wide checks from 16–20 April, 26 April - 1 May and 14–22 May found 142 wounded chicks (6.2% of the island fledgling count; Table 6.1). In addition to wounded chicks, 277 grey-headed albatross chick carcasses were found suggesting that mice attacked more than 18% of pre-fledging chicks in 2016 (combined wounded chicks and carcasses). On Grey-headed Albatross Ridge, 22 attack sites were found, containing 62 wounded chicks (8.8% of all chicks on the ridge), of which seven sites were further inland than in 2015 (Fig. 6.2). The area known as 'V-neck' had an estimated 53 affected chicks, substantially more than in 2015 (16 affected chicks). The number of wounded chicks and carcasses at all breeding sites increased dramatically towards the end of April through May (Fig 6.4). On the final check (14–22 May), 234 fresh (since 1 May) carcasses were counted (9% of the fledglings counted at the last check).

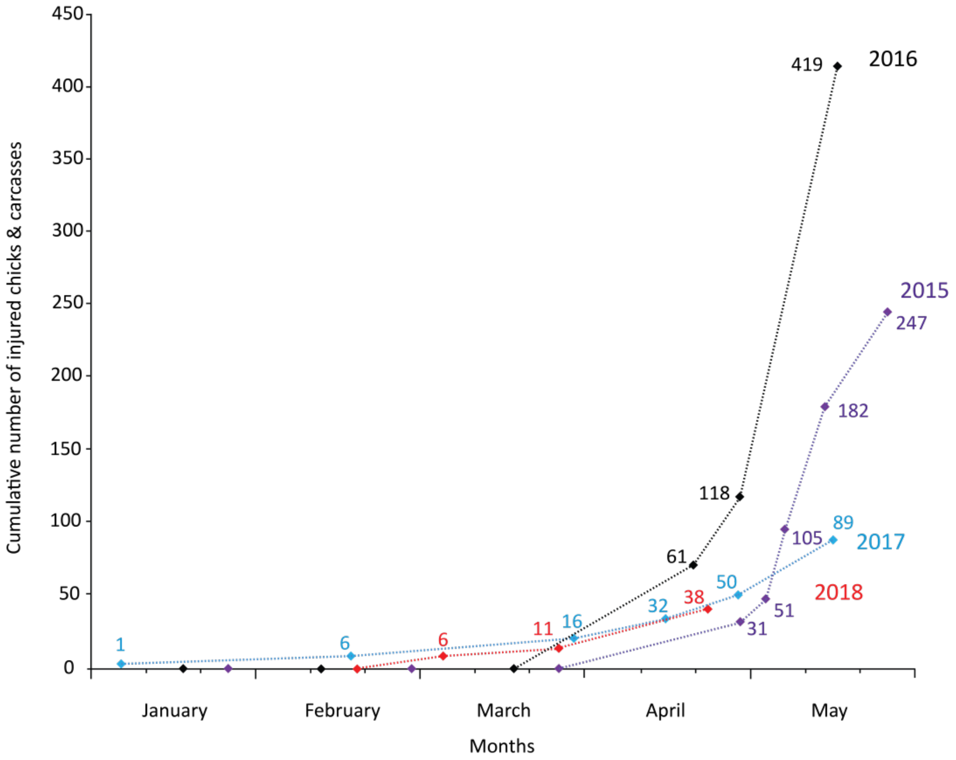

Figure 6.4. The number of mouse-injured and dead (carcass) grey-headed albatross chicks recorded at breeding sites at Marion Island in 2015–2018. The numbers at each data point represent the cumulative minimum number of wounded chicks and carcasses recorded at attack sites on each successive island chick count.

In 2017, an adult grey-headed albatross was seen brooding its mouse-injured chick on Grey-headed Albatross Ridge on 4 January (Fig. 6.5). This was the earliest sighting of a chick being attacked by mice, and the first of brooded chicks being attacked in this species. Subsequent island-wide checks from 16–28 February, 25 March - 4 April, 13–30 April and 16–20 May found 33 wounded chicks (at 24 sites, 1.4% of the island fledgling count; Table 6.1, Fig. 6.6). In addition, 56 carcasses where found, suggesting almost 4% of grey-headed chicks were affected in 2017. Attack sites were widespread on Grey-headed Albatross Ridge and along the coastal cliffs at Rook's, however Rook's Bay had substantially fewer affected chicks than in 2016.

Figure 6.5. An adult grey-headed albatross brooding its mouse-injured chick on Grey-headed Albatross Ridge at Marion Island on 4 January 2017. These were the earliest sightings of chicks being attacked by mice, and the first of brooded chicks being attacked in this species (photo Kim Stevens).

Figure 6.6. A well feathered grey-headed albatross fledgling with severe mouse-injuries on Grey-headed Albatross Ridge at Marion Island on 1 May 2017 (photo Kim Stevens and Christiaan Brink).

In 2018, 32 mouse-wounded chicks and six carcasses were found, suggesting 1.3% of grey-headed chicks were affected, the least since 2015. On Grey-headed Albatross Ridge, most of the affected chicks (10 of the 18 on The Ridge) were at one site; no injured chicks or carcasses were seen at the the 'V-neck' site.

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Sooty albatross

Following the observations of suspected mouse attacks on eight sooty albatross chicks at two sites in April 2009 (Jones and Ryan 2010), I found another wounded chick at the Toffee Lava nest site (Fig. 6.7a) on 29 May 2010. There were no further sightings of injured chicks until 2015, when a comprehensive survey of colonies around the island in April-May found wounded sooty albatross chicks at 14 of 104 colonies (13.5%, Fig. 6.7a, Table 6.2). Attack sites were 3.8 ± 3.9 km from the nearest adjacent attack site (range 0.1–10.7 km, *n* = 14).

Table 6.2. Estimated numbers of sooty albatross chicks attacked by mice on Marion Island from 2015–2018.

Counts	2015	2016	2017	2018
Number of chicks checked	1045	1072	734	1398
Minimum number of wounded chicks	45	50	28	15
Carcasses at or nearby attack sites	49	63	9	2
Carcasses away from attack sites	6			
Carcasses linked to mice	89% [a]			
Chicks attacked	4.2%	4.7%	3.8%	1.1%
Chicks affected (wounded + carcasses)	9.0%	10.5%	5.0%	1.2%
Number of attack sites/areas	15	21	13	8
Mean (SD) number of wounded chicks per site	3.5 ± 2.5%	3.8 ± 2.8%	3.1 ± 3.9%	2.3 ± 2.1

[a] % carcasses linked to mice excludes 15 chicks at one site killed by giant petrels during an extreme wind event (Dilley 2013a).

Of the 1,045 sooty albatross chicks checked in 2015, at least 45 chicks had mouse wounds (4.2%, 3.5 ± 2.5 chicks per attack site, range 1–8 chicks per colony). Similar to grey-headed albatross chicks, most sooty albatross wounds were on the crown, nape or back of the neck (96%, *n* = 45); only two sooty albatross chicks were attacked away from the head, with wounds on the elbow joints. Numbers of attacked chicks increased as winter set in; 32 wounded chicks were found at 12 colonies during the first survey (30 April - 6 May) and at least a further 13 during the second survey (15–26 May 2015), when two new sites were recorded: Triegaardt Bay South and Sealer's Beach (Fig. 6.7a). Wound progression and chick mortality varied among sites. At one site on the coastal cliffs below Lou-se-kop all four wounded chicks observed on 1 May were still alive on 5 May (three largely unchanged; one with a considerably enlarged wound). However, at Triegaardt Bay North, three of four wounded birds observed on 30 April had died by 5 May; the remaining chick's injury was more severe and two additional chicks had been attacked. On 30 April there were 23 fledglings and seven fresh carcasses at this site; two new carcasses were present on 5 May and a further 12 carcasses on 25 May, when only six fledglings remained (two of which had mouse wounds). This suggests that at least 21 chicks were killed by mice at this colony, which probably fledged fewer than 10 chicks from 160 pairs incubating in November 2014.

Overall, 70 fresh chick carcasses were found, of which 64 (91%) were at colonies with mouse wounded chicks, despite these colonies supporting <20% of chicks surveyed. However, 15 carcasses were found when exceptionally strong winds allowed giant petrels access (similar to behaviour documented in 2012; Dilley 2013a) to part of a large colony south of Triegaardt Bay on 4 May 2015, before the first mouse attacks were recorded at this site. Excluding these carcasses, 89% of carcasses were found at colonies where mouse attacks took place (Table 6.1). Combining the injured chicks (45) and mouse-related carcasses (49), it is likely that mice attacked ~9% (94/1045, Fig. 6.7a) of pre-fledging chicks in 2015, and that most of these chicks died. This is a conservative estimate because some early chick mortalities may have been missed and many carcasses could have fallen into the sea or been carried away by giant petrels or skuas.

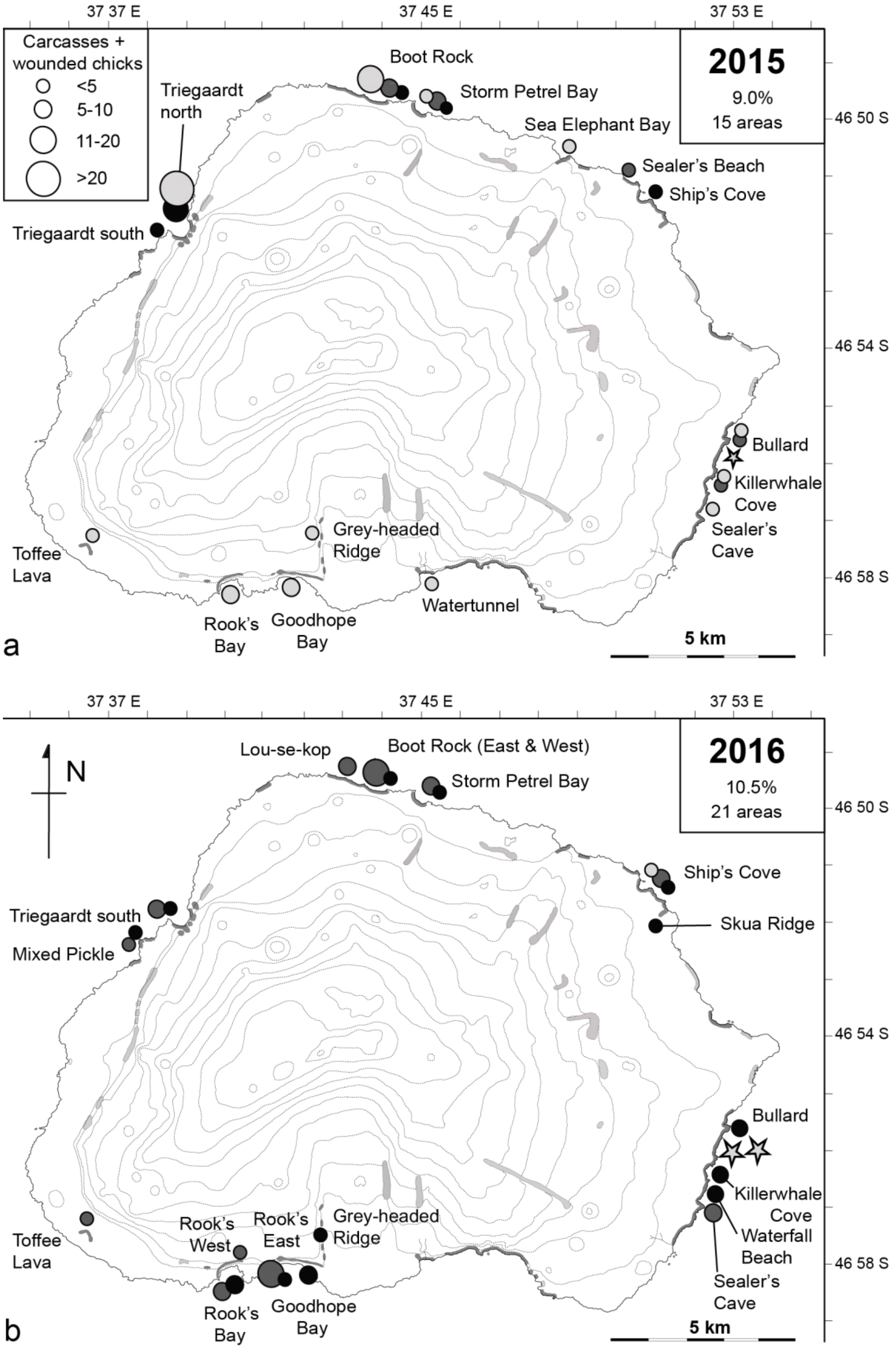
2016

All the breeding sites around the island were surveyed on four occasions in 2016. On the first survey in mid-March, 1,072 chicks were counted. One mouse injured chick was seen at Ship's Cove on 24 March with a large wound on the back of the neck extending to the scalp. This chick was still alive on 2 April, but the wound was more extensive and the chick had disappeared by mid-April.

During the second survey (15–20 April), 34 injured chicks were counted at 10 sites (3.4 ± 2.3 injured chicks per site), but no carcasses were seen. On the third survey one week later (29 April - 2 May), 37 injured chicks and 26 carcasses were counted at 16 sites (2.6 ± 2.4 injured chicks per site). Six of these sites were considered new attack sites since the second survey (~one week earlier): three were >1 km from the nearest affected site, but three were relatively close (<300 m) to an already affected site. The number of affected birds appeared to increase rapidly at most sites between surveys. For example, on the slopes of Rook's East the numbers of injured chicks compared to carcasses seen at each successive survey progressed from 5/0, to 8/2 to 10/2 from 17–30 April; and at Boot Rock East there were eight injured chicks and no carcasses on 19 April, but at the next check on 2 May, only four of the eight injured chicks were alive and there were five fresh carcasses. On the fourth survey (14–22 May), 11 injured chicks and 37 fresh carcasses were counted at 12 sites (5 of which were new sites for 2016). Overall, 50 wounded chicks and 63 carcasses were counted at 21 sites (Fig. 6.7b) in 2016. It is likely that mice attacked >10% (113/1072, Fig. 6.8) of pre-fledging chicks in 2016.

2017

Five surveys were completed in 2017. No wounded chicks or carcasses were seen on the first survey (16–28 February, 734 chicks). On the second survey (25 March - 4 April) five wounded chicks (of 15 chicks at site) were seen at one site on the slopes of Rook's East and 2 fresh carcasses (of 84 chicks at site) at Boot Rock. On the third survey (13–17 April), three of the five chicks at Rook's East were still alive but had more extensive wounds; single injured chicks were seen at two new sites near Killerwhale Cove; and single fresh carcasses were counted at three new sites (Puisie, Rook's East and Water Tunnel). No wounded chicks or carcasses were seen at the other sites.


37 37 E
37 45 E
37 53 E
Carcasses + wounded chicks
<5
5-10
11-20
>20
Boot Rock
Triegaardt north
Storm Petrel Bay
2015
9.0%
15 areas
46 50 S
Sea Elephant Bay
Sealer's Beach
Ship's Cove
Triegaardt south
46 54 S
Bullard
Killerwhale Cove
Sealer's Cave
Toffee Lava
Grey-headed Ridge
46 58 S
Rook's Bay
Goodhope Bay
Watertunnel
5 km
a
37 37 E
37 45 E
37 53 E
Lou-se-kop
Boot Rock (East & West)
Storm Petrel Bay
2016
10.5%
21 areas
N
46 50 S
Ship's Cove
Triegaardt south
Skua Ridge
Mixed Pickle
46 54 S
Bullard
Killerwhale Cove
Waterfall Beach
Sealer's Cave
Toffee Lava
Rook's West
Rook's East
Grey-headed Ridge
46 58 S
Rook's Bay
Goodhope Bay
5 km
b

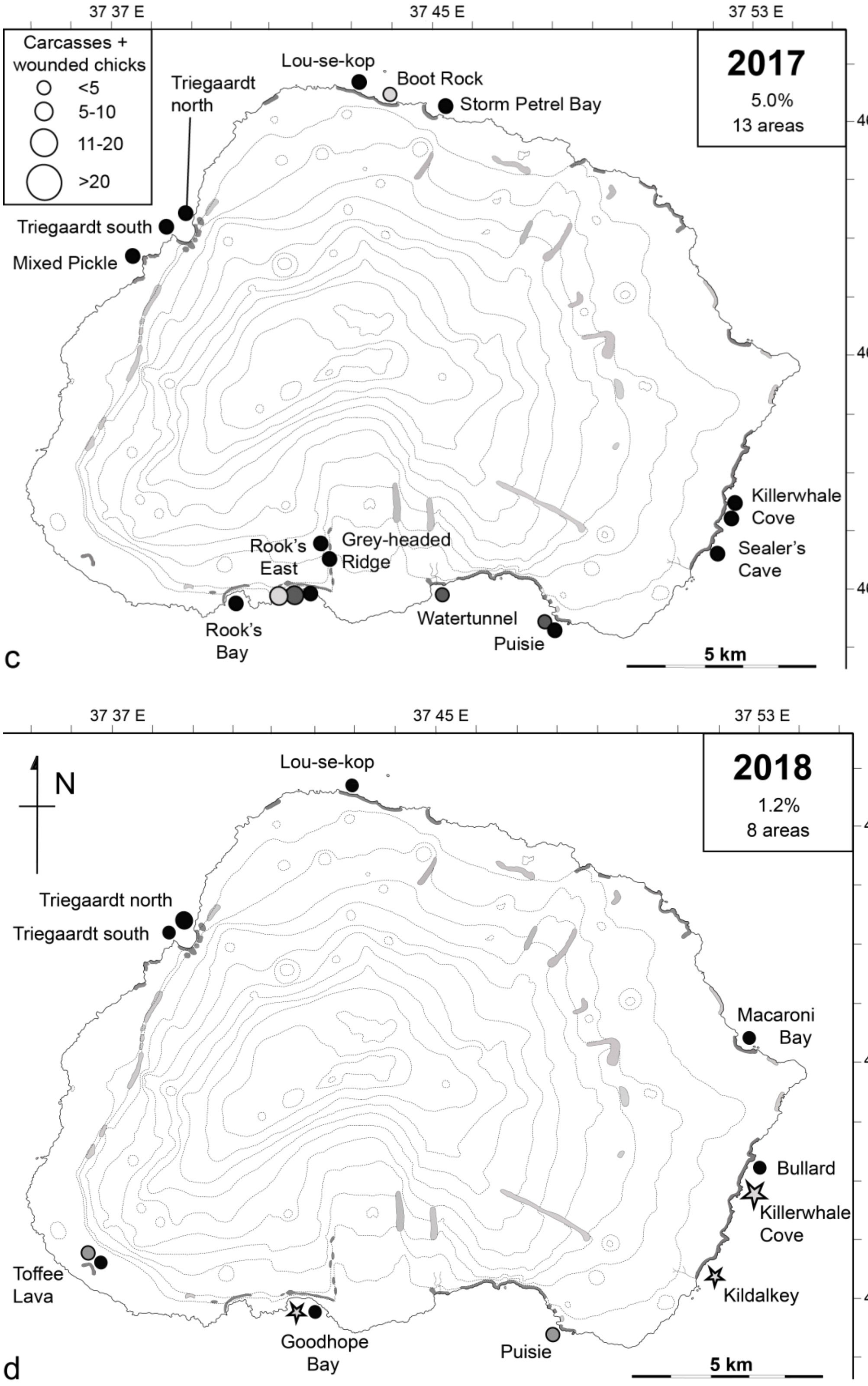


Figure 6.7. The locations of sooty and light-mantled albatross breeding colonies and mouse attack sites at Marion Island in 2015–2018. Light-mantled albatross breeding sites are light-grey; attack sites indicated by stars (small star = 1 wounded chick; large star = 2). Sooty albatross breeding sites are dark-grey; Numbers affected at attack sites indicated by shaded circles (see inset key). Conventions as in Fig. 6.2.

Three wounded chicks were counted at two new sites (Grey-headed Ridge (1) and south of Killerwhale Cove (2)) on the fourth survey (26 April - 1 May 2017), but no carcasses were seen. The three wounded chicks at the Rook's East site (seen on the previous two surveys) had disappeared with no sign of the carcasses. The fifth survey (14–23 May) had the highest counts of all five surveys: 13 wounded chicks (at seven sites) and four carcasses (at three sites). Of the 13 wounded chicks, three were new victims at the same Rook's East site as seen in the second and third surveys, and 10 were at seven new sites where wounded chicks had not been seen in 2016. Overall, a total of 28 wounded chicks and nine carcasses were counted at 13 sites (Fig. 6.7c) in 2017. It is likely that mice attacked ~5.0% (37/734) of pre-fledging chicks in 2017.

2018

On the first survey (28 February - 8 March) one wounded chick was seen at the Toffee Lava and on the second survey (25–30 March) one fresh carcass was counted at Puisie cliffs. On the final survey (19–25 April), 14 wounded chicks were counted at six sites, with the greatest number on the cliffs at Trichaardt North (7/68 chicks). Colonies were not surveyed in May 2018 (as in previous years) and as such the total number of affected chicks is not comparable with previous years. Overall, 16 wounded chicks and no carcasses were counted at seven sites (Fig. 6.7d) in 2018, affecting 1.1% (16/1398) of pre-fledging chicks.

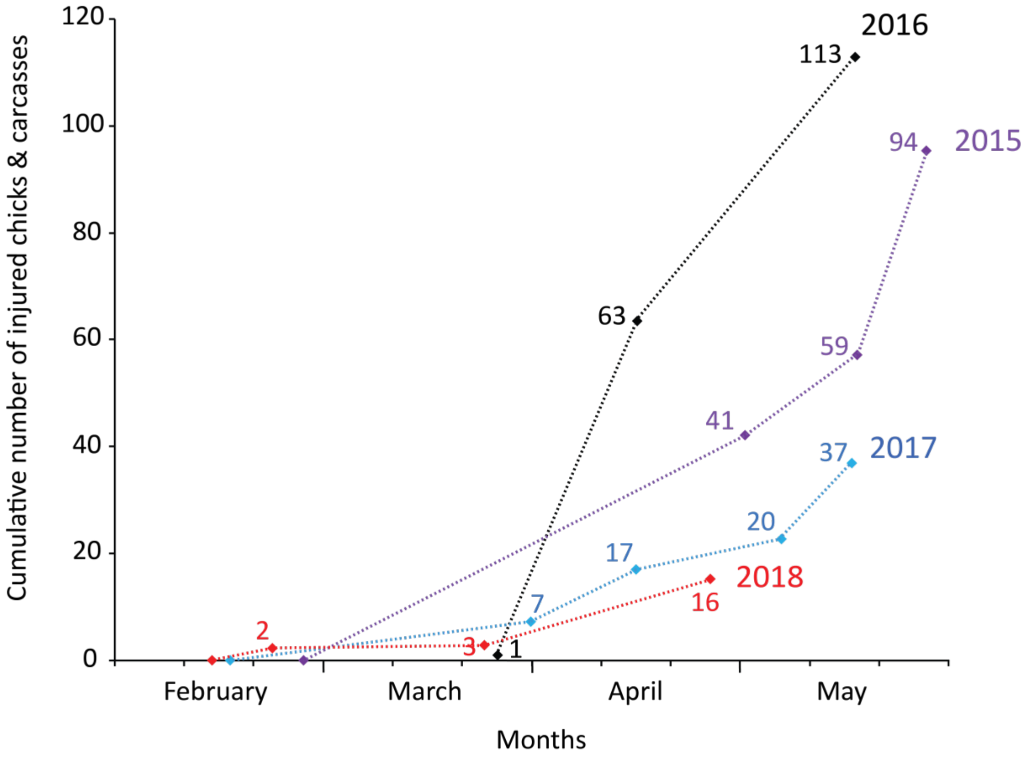


Figure 6.8. The number of mouse-injured and dead (carcass) sooty albatross chicks recorded at breeding sites along the 72 km of coastline at Marion Island in years 2015–2018. The numbers at each data point represent the cumulative minimum number of wounded chicks and carcasses recorded at attack sites on each successive island chick count.

Light-mantled albatrosses

The first wounded light-mantled albatross chick was found in 2015 among 25 surveyed at nine coastal and two inland breeding sites (4%); it was found on the east coast near Bullard South (Fig. 6.7a) at a mixed-species colony that also contained a wounded sooty albatross chick. Unlike most of the mouse-inflicted neck and scalp wounds seen on sooty and grey-headed chicks, this chick had a wound on the outer tail base. Wounded chicks were found in three of the four years surveyed (Table 6.3), however carcasses were not included in the counts as it was not possible to tell them apart from sooty albatross carcasses.

Table 6.3. Estimated numbers of light-mantled albatross chicks attacked by mice on Marion Island from 2015–2018.

Count	**2015**	**2016**	**2017**	**2018**
Number of chicks checked	25	65	27	98
Minimum number of wounded chicks	1	4	0	4
Areas with wounded chicks (see Fig. 6.7a, b, d)	1	2	0	3
% of chicks attacked	4.0%	6.2%	0%	4.1%

Wandering albatrosses

Since 2003, 32 wandering albatross chicks have been found with mouse wounds in the three study colonies (average 2.1 ± 2.0 per year, range 0–6, 272 ± 42 nests monitored per year, Figs. 6.9 and 6.10), with 16 incidental sightings from other parts of the island (Fig. 6.10). Twenty-five of the 32 attacks (78%) were first observed in June (*n* = 8), July (*n* = 9) and August (*n* = 8), but mouse-wounded chicks were seen from 6 April to 11 November. Of the chicks that were checked repeatedly, 22 of 32 (69%) died from their wounds, or when attacked by giant petrels; 10 (31%) recovered from their wounds and fledged.

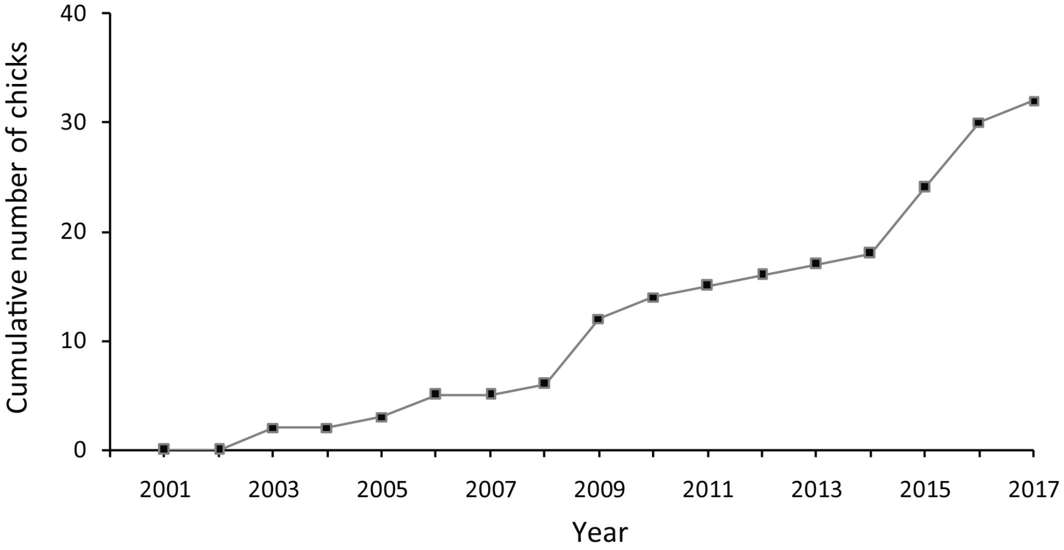


Figure 6.9. The cumulative number of mouse-wounded wandering albatross chicks in the three *study colonies* from 2001 to 2017.

Of the 16 attacks recorded in 2015 (the largest number of attacks over the monitored years), six were within the study colonies (five at Sealer's Beach and one at Macaroni Bay) and 10 were recorded around the rest of the island (Fig. 6.10). A survey of chicks outside study colonies from 7 June to 2 July 2015 inspected 749 chicks (of 850 fledglings island wide (excluding the 191 in the study colonies)), of which five (0.7%) had mouse wounds: one near Kampkoppie, one near Kaalkoppie and three near Swartkop Point, all on the island’s west coast. Subsequent visits to Swartkop Point found these five chicks had disappeared (by 20 July) and another six injured chicks where found at a cluster (~200 m diameter) of nearby (<500 m) nests, which were all empty on 15 October 2015. Prior to 2015, mice targeted the rump (n = 21), wing (3) or shoulder (1), but in 2015 three of the 16 attacks were head wounds (Fig. 6.11).

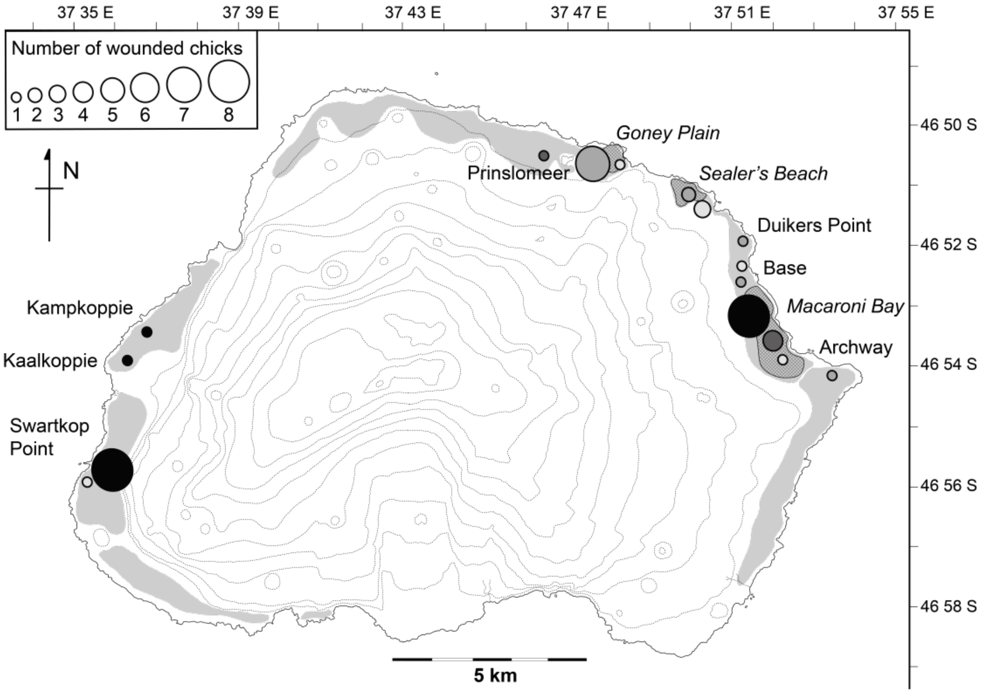


Figure 6.10. The distribution of mouse-wounded wandering albatross chick sightings from 2003–2018 at Marion Island. Observation effort was higher in the three study colonies (in italics, indicated with a hatch pattern) compared to breeding sites around the rest of the island (grey shaded areas). The numbers of wounded chicks were pooled into four groups of four years: 2003–2006 (light grey shade inside circle), 2007–2010 (medium grey), 2011–2014 (dark grey) and 2015–2018 (black). The size of the circle at each attack site indicates the sum of the wounded chicks over each four year period.

Figure 6.11. A mouse feeding on the crown of a wandering albatross chick at Sealer's Beach study colony, Marion Island, 15 June 2015. This chick died five days later (photos Stefan Schoombie).

Infra-red trap cameras and frequent nest checks were used to monitor the progress or fate of injured chicks. Images captured on these cameras confirmed that mice were responsible for the wounds typically seen on chicks: rump wounds (in July 2013 at Macaroni Bay and June 2014 at Sealer's Beach);

elbow/wing wounds (July 2016 at Macaroni Bay); and crown/scalping wounds (in June 2015, Fig. 6.11). Here I describe events from two nests.

In May 2015, trap cameras captured images of up to four mice at a time on a mouse-wounded chick. Mice were photographed persistently nibbling at the wound on the chick's lower rump (to the base of the tail feathers) which appeared to keep the chick awake through much of the night (Fig. 6.12). The chick survived for multiple nights but appeared very weak and was finally killed by giant petrels.

Figure 6.12. An infra-red remote camera captured up to four mice feeding at one time on the lower rump of this wandering albatross chick (left), causing a large wound (right). The chick survived for multiple nights but was finally killed by giant petrels (photos Stefan and Janine Schoombie, Marion Island, 25 June 2015).

On 19 June 2016, a three month old chick in the Macaroni Bay study colony was first noted with oil on its face and neck. At subsequent weekly checks the chick looked progressively weaker with very wet and oily down, especially on its neck and head, but was uninjured on 20 July. By 28 July, the chick had extensive wounds on its elbow, and it disappeared a week later (Fig. 6.13).

Figure 6.13. A four month-old wandering albatross chick with extensive mouse-wounds on its elbow and lower rump above the tail. This chick disappeared less than two weeks after it was first seen with injuries (photo Kim Stevens and Christiaan Brink, Marion Island, 28 July 2016).

Discussion

Prior to 2015, mouse attacks on albatross chicks have only been confirmed to occur on Gough Island (Wanless *et al*. 2009; Appendix 3) and suspected on Marion Island (Jones and Ryan 2010). The observations reported here confirm that mice are responsible for the wounds observed on all three albatross genera breeding on Marion Island, and that large numbers of grey-headed and sooty albatross chicks died from these wounds in 2015. This is the first direct evidence that mice are responsible for fatal attacks on surface-nesting seabird chicks at Marion Island. Given the presence of field researchers on Marion since the 1980s and the striking nature of the wounds inflicted, it is unlikely that mouse attacks on surface-nesting seabirds were overlooked prior to the first records in 2003. These are also the first records of extensive mouse predation on *Thalassarche* and *Phoebetria* albatross chicks. Although mice are known to be serious predators of Tristan albatrosses on Gough Island (Wanless *et al*. 2009; Appendix 3), there have been only two records of mice killing chicks of these summer-breeding albatrosses: one sooty albatross and one Atlantic yellow-nosed albatross *Thalassarche chlororhynchos* (Cuthbert *et al*. 2013b).

Mice are the only introduced mammals at both Gough and Marion Islands. Wanless *et al*. (2007, 2009) hypothesized that the impacts of mice on seabirds are most severe on such islands because mouse populations are not regulated by the effects of dominance, competition and predation by other, larger introduced mammals (e.g. cats or rats). On Marion Island, mice were not an important food source for cats (van Aarde 1980) so it is unlikely that cat predation limited the mouse population (van Aarde *et al*. 1996). However, cats may have influenced mouse demography, and their removal, combined with a warming climate (Le Roux *et al*. 2008), may have allowed mouse densities to increase (Ferreira *et al*. 2006). In 2008–11, densities in mire habitats on Marion Island reached up to at 237 mice·ha^{-1} (McClelland 2013), similar to peak densities on Gough Island (266 mice·ha^{-1}, Cuthbert *et al*. 2016).

A striking feature of the attacks on Marion Island albatrosses since 2015 was that most wounds were on the chicks' heads and necks. Mice on Gough Island mainly attack albatrosses on the rump or wings (Wanless *et al*. 2009; Appendix 3), and prior to 2015, all wounds on wandering albatrosses on Marion Island also were on the rump or wings (Jones and Ryan 2012). Most mouse attacks on seabird chicks at both islands take place in winter, when mouse populations crash as food resources are depleted (Matthewson *et al*. 1994; Cuthbert *et al*. 2016). *Thalassarche* and *Phoebetria* albatross chicks fledge in autumn or early winter, and the chicks are thus quite mobile and presumably better able to fight off mice by this time of year, and perhaps even more importantly, they are also well feathered. I hypothesise that the dense cover of long contour feathers prevents mice from attacking the rump area where they usually target downy *Diomedea* albatross chicks.

Mice gain two benefits by targeting albatross crowns and napes: they are safe from retaliation by the chick's bill, and the short feathers on the crown make it easier to reach the skin. This novel attack technique allows mice on Marion Island to attack fully-feathered chicks, making available the chicks of summer-breeding species in the critical period as food resources dwindle in April-May. Gough Island mice had not yet learned this behaviour in 2015; the few *Thalassarche* and *Phoebetria* albatross chicks attacked on Gough Island were killed as downy chicks in December-January by mice entering the nest cup from below (Cuthbert *et al*. 2013b). When it was first discovered that house mice were significant predators of seabirds on Gough Island, much was made of the fact that they are larger (average adult body mass 35 g) than any other island mouse population (Cuthbert and Hilton 2004; Wanless *et al*. 2007; Cuthbert *et al*. 2016). This might confer an advantage in subduing petrel chicks (Chapter 4), but the observations reported here from Marion Island show that large body mass is not a prerequisite for attacking large albatross chicks. Adult body mass of mice on Marion Island (21 g; Avenant and Smith 2003) is similar to mice on other islands (e.g. 19–22 g at Antipodes, Russell 2012; 21 g at South Georgia, Cuthbert *et al*. 2012), and has not increased since cats were eradicated (Ferreira *et al*. 2006; McClelland 2013). Indeed, smaller size might make it easier for mice to cling onto the heads of albatross chicks. The fact that 'normal' house mice are able to attack and kill large albatross chicks indicates the need for vigilance wherever mice have been introduced to seabird breeding islands.

It is tempting to speculate what might have triggered the sudden increase in mouse attacks on albatrosses at Marion Island in 2015 (compared to the scattered incidents in previous years) and the even bigger increase in 2016. One hypothesis is prey switching by mice, whereby mice supplement the invertebrate component of their diet with seabird chicks, driven by a steady decrease in invertebrate biomass on Marion Island over the last 40 years (Burger 1978; Gleeson and van Rensburg 1982; Smith *et al*. 2002; McClelland 2013). Recent evidence of the first extensive plant die-back and drying out of mire lakes and peats as a result of atmospheric warming and reduced precipitation on Marion Island (Hedding and Greve 2018) could have knock-on effects to the invertebrate communities and mouse population dynamics (resulting in unusually high densities in late autumn). For example, Marion Island's precipitation has fallen from an average of 2,727 mm in the 1960s to an average of 1,778 mm (2007–2016), and 2015 is the driest year on record (1,550 mm; D. W. Hedding pers. comm.) since 1949.

Another intriguing question is how attacks on albatross chicks commenced at scattered locations all around Marion Island's ~72 km coastline. Multiple attacks typically occurred at each affected site, suggesting some cultural transmission of this novel foraging technique (cf. Wanless *et al*. 2005), but this mechanism cannot explain how attacks were initiated seemingly independently at sites separated by distances hundreds of times greater than mouse home ranges (Cuthbert *et al*. 2016). Among animal populations, once an individual in a population inovates a novel behaviour, it is often rapidly transferred to other individuals (Laland and Janik 2006). It is thought the mechanism behind this transfer of behaviour is through observation or imitation, but whether this behaviour moves between successive generations (cultural transmission, Whiten *et al.* 2011) of mice of Marion Island is unknown, although it is plausible for mice which survive through the winter.

Croxall *et al*. (2012) listed 73 priority islands where the eradication of invasive alien vertebrates would benefit globally threatened seabirds. House mice are present on 25 of these islands; five have no other invasive vertebrates, and of these, only Gough Island supports breeding albatrosses. If the high levels of mouse predation recorded on sooty and grey-headed albatrosses at Marion Island in 2015 and 2016 recur in the coming years, they may have long-term demographic consequences on these populations, suggesting that Marion Island is a priority island for mouse eradication and should be added to this list.

Chapter 7

Synthesis of key results and the way forward

Gough Island viewed from beneath a helicopter equipped with a bait bucket during the non-toxic bait trials in 2013 (photo snapshot from a GoPro, Ben Dilley).

The catastrophic impacts of invasive mammals on seabird populations have been extensively studied over the last few decades (e.g. Atkinson 1985; Jones *et al*. 2008) and are a global conservation concern. In many cases the introduction of alien mammals has radically altered island ecosystems (Mulder *et al.* 2011). The research presented in this thesis forms part of a larger drive to assess whether the impact of mice on seabirds alone is sufficient to warrant eradication campaigns.

My aim in this thesis was to document the impacts of invasive mice at Gough and Marion Islands and to establish pre-eradication baselines for the petrel populations on Marion Island. In this synthesis chapter I pull together the key results of the thesis, provide further discussion on subjects which I was only able to touch on briefly in the main chapters, and highlight areas for future research. I also consider the way forward, by providing an update on plans to rid both Gough and Marion Islands of mice.

A brief synthesis of the key results

1. Systematic petrel surveys are possible and worthwhile

The systematic surveys provided the first reasonably accurate population estimates for white-chinned and blue petrels breeding at Marion Island. These island-wide systematic surveys involved extensive walking of all possible breeding habitats around the island. As a novice fieldworker to oceanic islands and burrow-nesting petrels in 2009, the white-chinned petrel survey was an invaluable experience to realise what can be done – although the survey required some effort, the resulting error margins are relatively small. The blue petrel survey required an even bigger survey team and extensive post-survey occupancy trials in order to gain an understanding of the occupancy rates around the island in different habitats with different burrow densities. Interestingly, the early population estimates from Williams (1979) in the mid-1970s were not too far off (white-chinned and blue petrels were each estimated at '10s of 1,000s of breeding pairs' for Marion), but tend to underestimate populations at least for burrow-nesting petrels (compared to the more accurate counts of surface-nesters).

The transect survey which was completed during the relief voyage in 2015 was a more efficient survey method, but only a tiny fraction of the island area was covered resulting in wide error margins around the resultant population estimates. As discussed in Chapter 2, multiple factors affect which survey method is most appropriate for a particular species and island, but where possible, a systematic survey is worthwhile to be able to detect long-term changes in populations, especially post-eradication of invasive mammals. One of their key values is a greater ability to detect changes in range (i.e. area of island occupied), since population sizes vary through changes in range and density, and random transects are less sensitive to changes in the former than are systematic surveys.

2. Marion's burrow-nesting petrel populations are recovering slower than expected post-cats

My repeat survey in 2013 showed that petrel burrow densities at Marion have increased by a modest 56% since 1979. I was fortunate to have continuity in approach between the two surveys as Mike Schramm, who made the 1979 counts, returned to Marion in 2012 to ensure that the areas sampled were matched as closely as possible. Interestingly, before I even completed the repeat survey, the general perception among experienced biologists who have been involved with work at Marion since the cat-era, was that the burrow-nesting petrel numbers had not recovered as expected and birds were still not nearly as common as the numbers seen at night in the 1970s and early 1980s (pers. comm. Valdon Smith, Niek Gremmen, Mike Schramm, Marthan Bester, John Cooper). Recent evidence of a steady decrease in the number of skuas breeding on Marion Island, which are major predators of burrowing petrels, also points towards a slow recovery of burrow-nesting petrel populations (Ryan *et al.* 2009).

3. First visual and quantitative evidence of mouse impacts on Marion's burrow-nesting petrels

Mice are suppressing the recovery of burrow-nesting petrel populations on Marion Island, especially those that breed in winter, through depredation of eggs and chicks. Although predation rates appear to be lower than on Gough Island, I did eventually succeed in obtaining video evidence of mice attacking burrow-nesting petrel chicks at Marion Island. I consider the combined evidence from mouse visitation rates inside video-monitored burrows and the breeding success data over five successive seasons to be sufficient evidence to conclude mice are suppressing the recovery of petrels at Marion Island.

4. First records of mouse impacts on Marion's summer breeding albatrosses

The unprecedented increase in 2015 and 2016 of mice attacks on large chicks of all three albatross species that fledge in autumn was widespread and unexpected. In 2017 and 2018 fewer albatross chicks were attacked, but reasons for these fluctuations are not clear and warrant further study. These attacks highlighted how summer breeding species are also vulnerable to the impacts of mouse predation - impacts which are likely to increase as climate change promotes higher mouse densities.

5. Visual and quantitative evidence of mice impacts on multiple seabird species on Gough

Mice were thought to have a negative effect on multiple seabird species, yet earlier studies on Gough Island had confirmed predation on only a few species. Although my observations were recorded over a single summer/winter season, the results expand on the findings of Richard Cuthbert (2000/01) and Ross Wanless (2003/04) on Gough Island. Using the underground burrow cameras I confirmed that mice depredate six species of burrow-nesting petrels. In addition, the quantitative video evidence of attack rates on Tristan albatross chicks (using motion-activated cameras of 20 chicks in the Gonydale study area) revealed 14/15 failures (93%) were due to mouse predation and that predation events were often rapid, taking only a few nights. This enlightening result explained how seemingly healthy chicks 'disappeared' between weekly colony checks.

When did mice start depredating seabird chicks?

It is unclear when mice started attacking seabird chicks at Marion Island as the timeline is complicated by the presence of cats as the superpredators from 1949–91, but it is likely to have occurred at least since the early 1980s. Mice have been present on Marion since the early 1800s and biological researchers have been monitoring some seabird species year-round since 1965 and more intensively since the 1980s and 1990s (Cooper *et al*. 2001). Surface-nesting seabirds such as albatrosses are well studied, as these species are readily observed and are therefore easier to monitor; the first wandering albatross *Diomedea exulans* chick injured by mice was found in 2003 (Jones and Ryan 2010). There are few early records of burrow-nesting petrel populations, but the destructive impacts of cat predation were well documented (Van Aarde 1980). Michael Schramm (personal communication 2017) found no evidence of mouse predation on live or dead burrow-nesting petrel chicks during his intensive monitoring of 137 *Pterodroma* burrows over 14 months in 1979–80 (Schramm 1983), but Fugler *et al*. (1987) found evidence of blue petrel chicks injured by mice at Long Ridge in 1982.

On Gough Island the behaviour of mice depredating seabird chicks was only discovered for the first time in 2001, when the first seabird biologist spent a year on the island. However it could have been going on well before then on Gough, but remained undetected since the over-wintering members focused on weather observations. Since 2001, predation rates appear to have escalated. The data for Marion are more compelling – certainly the attacks on summer breeding albatross chicks are a new phenomenon (Chapter 6). While mouse-injured chicks in burrows could easily go undetected, it seems likely that albatross chicks with bloody mouse-inflicted wounds would have been noticed, especially on Marion Island where year-round fieldworkers have spent significant amounts of time in the field since the early 1980s. Some wandering albatross attacks on Marion might have been overlooked, especially as attacks remain rare, although mice numbers were probably constrained to some extent by cats until the late 1980s.

What may have triggered the sudden increase in mouse predation?

This is an intriguing question because mice have been present on both Marion and Gough Islands for around 200 years. As mentioned in Chapter 6, we can only speculate what might have triggered the sudden increase in mouse attacks at both study islands. There are two likely hypotheses to explain this sudden change in mouse behaviour: (1) prey switching by mice, whereby mice supplement the invertebrate component of their diet with seabird chicks, driven by a steady decrease in invertebrate biomass on Marion Island over the last 40 years (Burger 1978; Gleeson and van Rensburg 1982; Smith *et al*. 2002; McClelland *et al*. 2018); and/or (2) climate change, where the gradually warmer and especially drier climate (le Roux and McGeoch 2008; Hedding and Greve 2018) potentially allows for an extended mouse breeding season and higher mouse survival rates during milder winters. There is evidence of an increase in peak mouse densities on Marion Island between 1990 and 2008, yet invertebrate biomass has decreased >80% since the late 1970s (McClelland *et al*. 2018), suggesting this food resource is now in very short supply. If invertebrate biomass continues to decline, mice will likely target alternative food sources and the impact of mouse predation on Marion's seabird chicks is likely to become even more serious. This is the pattern that has been observed recently on Marion and Gough Islands since 2015 where there has been a marked increase in the frequency of mice attacking surface-breeding seabird chicks (Marion – Chapter 6; Gough – Jaimie Cleeland pers. comm. 2018), as well as the first attacks on incubating adults (Tristan albatross on Gough and northern giant petrel on Marion).

The impact of mice in the presence/absence of other invasive mammals

In a broader theoretical context, this thesis investigated how seabirds on Marion have responded to the removal of feral cats (top predator or superpredator), and how the mice (mesopredator) as the sole remaining introduced mammal, have responded to the removal of cats. The ecological consequences of removing only the superpredator are difficult to predict, especially when complex competitor/predator-prey dynamics exist and removal of one pest may not necessarily result in a simple or rapid reversal towards a pristine island. Removal of top predators may cause mesopredator release, resulting in significant and often inadvertent consequences (Courchamp *et al*. 2003). The ability to predict possible outcomes to well-intended management interventions may often be low, especially when several species are involved (Montoya *et al*. 2006). On sub-Antarctic Macquarie Island the eradication of feral cats by 2001 caused rabbit *Oryctolagus cuniculus* numbers on the island to increase significantly which led to substantial local and landscape-scale changes in vegetation (Bergstrom *et al*. 2009a; Terauds *et al.* 2014). Bergstrom *et al*.'s (2009a) study demonstrated that cats on Macquarie Island were exerting top-down control on the rabbit population, and that the eradication of the cats led to a substantial increase in rabbit numbers and an associated trophic cascade. Dowding *et al.* (2009) disagreed with these findings for various reasons, but primarily that a reduction in the application of the rabbit control agent, *Myxoma* virus, coinciding with cat removal, was a major driver of rabbit population release. Although controversial, these studies highlight how systems with multiple invasive species represent complex situations that require careful scrutiny which ideally should occur in advance of, during, and following management interventions (cf. Bergstrom *et al*. 2009; Zavaleta *et al.* 2001).

Had cats not been successfully eradicated from Marion in the 1980s, the petrel populations today would almost certainly be dramatically reduced and many species would very likely have disappeared completely, similar to the situation on Amsterdam Island where continued predation pressure from cats has reduced some petrel species to a only few pairs (Micol and Jouventin 1995). Mice significantly disrupt the functioning of terrestrial ecosystems on seabird islands (Chown and Smith 1993) and on Marion this has been happening for around 200 years (Berry *et al*. 1978). The mice may be seen as part of a syndrome of interacting factors (Parkes 2016) having adverse impacts on native invertebrates, plants and seabirds (e.g. Phiri *et al*. 2009; Angel and Cooper 2012). The dramatic decrease in burrowing petrel populations at Marion Island caused by the cats is presumed to have adversely affected key ecological processes driven by burrowing petrels such as soil turn-over and marine nutrient imports (Caut *et al*. 2012). The combinations of these mouse-driven changes have altered the state of Marion Island's ecosystems compared with the near-pristine condition of neighbouring mouse-free Prince Edward Island.

For more than 30 years, the burrowing petrel populations on Marion Island were impacted by both cats and mice. As discussed in Chapter 3, cat predation was far more detrimental than mouse predation to petrel populations because cats killed adults as well as chicks, affecting adult survival and reproduction, with much greater demographic consequences (Le Corre 2008). How would the presence of a second mesopredator, for example rats, have influenced the dynamics on Marion Island following the removal of cats? Would the removal of the top predator have triggered a 'mesopredator release' of both mice and rats? On other islands where rats and mice co-occur, the rats are thought to suppress mouse numbers (MacKay *et al*. 2007; Russell 2011; Brown *et al*. 1996; Weihong *et al*. 1999), but interactions between rats and mice are poorly understood and there is likely to be an element of both competition and predation (cf. Caut *et al*. 2007). Rats (100–150 g, Clark 1980) are much larger than mice (20–30 g, Avenant and Smith 2003), which allows rats to kill petrel chicks outright, rather than gradually over a period of a few nights. However it appears that aside from body mass, the difference is also in the method of dispatch, where a mouse may nibble a chick to death (Chapters 4–6), but rats actively predate. For example, Brooke *et al.* (2010) recorded how Pacific rats *Rattus exulans* actively seized and ate entire Murphy's petrel *Pterodroma ultima* chicks, resulting in heavy and total chick losses immediately after hatching in study years 1991 and 2003 at Henderson Island, central South Pacific. The significant difference in body mass between rats and mice could permit rats to dominate and increase in numbers following the removal of cats (Brown *et al*. 1996). On Hauturu Island (Little Barrier Island), off New Zealand, the removal of cats allowed Cook's petrel *Pterodroma cookii* numbers to recover after years of predation (Imber *et al*. 2003b) and interestingly the population of Pacific rats did not show any increase (Girardet *et al*. 2001).

How long can seabird populations sustain mouse impacts?

The widespread increase in mouse predation at both Marion and Gough Islands over the last decade is cause for concern. Left uncontrolled, how long can these seabird populations sustain current levels of mouse predation? It is feared that 21 of the 25 species breeding on Gough Island (Table 7.1) and 19 of the 29 species breeding on Marion Island (Table 7.2) may be vulnerable to local extirpation, should mice not be eradicated (figures are based on expert field knowledge of the severity of impacts, see Preston *et al*. 2017).

Table 7.1. Estimated risk of local extirpation of bird species currently known or thought to breed on *Gough Island* if the mice are not eradicated. Species in bold are **endemic** to Gough Island.

Species	**Estimated breeding pairs**	**Considered vulnerable to predation**	**Estimated years to local extirpation**
Black-bellied storm petrel *Fregetta tropica*	<5 000[1]	yes*	30
White-bellied storm petrel *Fregetta grallaria*	<5 000[1]	yes*	30
Grey-backed storm petrel *Garrodia nereis*	<5 000[1]	yes*	30
White-faced storm petrel *Pelagodroma marina*	<10 000[1]	yes*	30
MacGillivray's prion *Pachyptila macgillivrayi*	2 000[2]	yes	30
Common diving petrel *Pelecanoides urinatrix*	20 000[1]	yes	30
Great-winged petrel *Pterodroma macroptera*	<100[3]	yes	30
Gough bunting *Rowettia goughensis*	**1 000[1]**	**yes**	**30**
Atlantic petrel *Pterodroma incerta*	**900 000[4]**	**yes**	**50**
Grey petrel *Procellaria cinerea*	20 000[3]	yes	50
Little shearwater *Ardenna assimilis*	<10 000[1]	yes	50
Blue petrel *Halobaena caerulea*	200[5]	yes	50-100
Broad-billed prion *Pachyptila vittata*	1 500 000[1]	yes	50-100
Kerguelen petrel *Aphrodroma brevirostris*	20 000[1]	yes	50-100
Great shearwater *Ardenna gravis*	1 000 000[1]	yes	50-100
Soft-plumaged petrel *Pterodroma mollis*	400 000[1]	yes	50-100
Tristan albatross *Diomedea dabbenena*	**2 000[1]**	**yes**	**50-100**
Atl. yellow-nosed alb. *Thalassarche chlororhynchos*	5 000[1]	yes	50-100
Sooty albatross *Phoebetria fusca*	5 000[1]	yes	50-100
Brown noddy *Anous stolidus*	200[1]	yes	50-100
Antarctic tern *Sterna vittata*	500[1]	yes	50-100
Gough moorhen *Gallinula comeri*	3 500[1]	uncertain	
Southern giant petrel *Macronectes giganteus*	250[1]	uncertain	
Brown skua *Catharacta antarctica*	1 000[1]	uncertain	
Northern rockhopper penguin *Eudyptes moseleyi*	45 000[6]	no	

*Active burrows are very scarce, but previously these species were common on the island

Data sources: [1]Ryan 2007; [2]Ryan *et al*. 2014; [2]FitzPatrick Inst. unpubl. data; [4]Rexer-Huber *et al*. 2014; [5]Ryan *et al*. 2015; [6]Cuthbert *et al*. 2009

Table 7.2. Estimated risk of local extirpation of bird species currently known or thought to breed on *Marion Island* if the mice are not eradicated.

Species	**Estimated breeding pairs**	**Considered vulnerable to predation**	**Estimated years to local extirpation**
Grey-backed storm petrel *Garrodia nereis**	? [1]	yes – (but possibly already locally extirpated)	
Black-bellied storm petrel *Fregetta tropica**	? [1]	yes – (but possibly already locally extirpated)	
Grey petrel *Procellaria cinerea*	800[2]	yes	30
Cape petrel *Daption capense*	<5[2]	yes	30
Kerguelen petrel *Aphrodroma brevirostris*	5 000[2]	yes	50
South Georgian diving petrel *Pelecanoides georgicus*	1 000[1]	yes	50
Common diving petrel *Pelecanoides urinatrix*	2000[2]	yes	50-100
Great-winged petrel *Pterodroma macroptera*	14 000[2]	yes	50-100
Light-mantled albatross *Phoebetria palpebrata*	300[3]	yes	50-100
Sooty albatross *Phoebetria fusca*	1 465[3]	yes	50-100
Grey-headed albatross *Thalassarche chrysostoma*	7 900[1]	yes	50-100
Wandering albatross *Diomedea exulans*	1 800[1]	yes	50-100
Fairy prion *Pachyptila turtur*	1 000[1]	yes	50-100
Salvin's prion *Pachyptila salvini*	150 000[2]	yes	50-100
Blue petrel *Halobaena caerulea*	145 000[4]	yes	50-100
Soft-plumaged petrel *Pterodroma mollis*	5 000[1]	yes	50-100
White-chinned petrel *Procellaria aequinoctialis*	24 000[5]	yes	50-100
Antarctic tern *Sterna vittata*	25[1]	yes	50-100
Kerguelen tern *Sterna virgata*	50[1]	yes	50-100
Southern giant petrel *Macronectes giganteus*	1 750[1]	uncertain	
Northern giant petrel *Macronectes halli*	400[1]	uncertain	
Crozet shag *Leucocarbo melanogenis*	270[1]	uncertain	
Brown skua *Catharacta antarctica*	300[6]	uncertain	
Kelp gull *Larus dominicanus*	100[1]	uncertain	
Lesser sheathbill *Chionis minor*	700[1]	uncertain	
King penguin *Aptenodytes patagonicus*	220 000[1]	no	
Gentoo penguin *Pygoscelis papua*	900[1]	no	
Macaroni penguin *Eudyptes chrysolophus*	370 000[1]	no	
Southern rockhopper penguin *Eudyptes chrysocome*	67 000[1]	no	

*Current breeding not proven but suspected

Data sources: [1]Ryan & Bester 2008; [2]FitzPatrick Inst. unpubl. data; [3]Schoombie *et al*. 2017; [4] Appendix 1; [5] Appendix 2; [6]Ryan *et al*. 2009b.

Plan of action

In many cases, once invasive species are well established on large land masses they are impossible to eradicate; ongoing control is the only effective mitigation measure. Fortunately, eradicating introduced rodents from islands is a viable option using poison bait, either hand baiting (small islands) or aerial baiting (large islands; MacKay *et al*. 2007). Techniques for eradicating rodents from islands using aerial baiting have been developed since the early 1990s and there has since been a significant development and application of this conservation tool (Howald *et al.* 2007). By the late 1990s rodent eradications had been completed on two islands over 1300 ha in size (Kapiti Island off the south-west coast of New Zealand's North Island and Whenua Hou/Codfish Island off the west coast of Stewart Island; cf. Cromarty *et al.* 2002). New Zealand's Department of Conservation identify three core principles to achieve eradication success: (i) every reproductive individual mouse can be put at risk by the eradication technique(s), (ii) mice must be killed at a rate exceeding their rate of increase at all densities, and (iii) strict biosecurity measures must be in place (i.e. immigration must be zero) (Cromarty *et al.* 2002). However, strong evidence is vital to initiate and secure support for large-island operations, which are expensive and logistically challenging to conduct. Plans to eradicate mice from both Gough and Marion islands are underway, with eradication attempts planned for Gough Island in winter (May-July) 2020 and Marion Island in winter 2021.

Considering both islands' importance as breeding sites for threatened albatrosses and other seabird species that are being killed by mice (Tables 7.1 and 7.2), there is an urgent need to eradicate mice. Detailed feasibility plans for both islands (Parkes 2008, 2016) suggest that mice can be eradicated using aerial baiting. This follows the now well-established approach of using helicopters fitted with GPS guidance systems and under slung bait-distribution buckets to spread brodifacoum-laced pellets across the entire island over a relatively short period, to ensure that all rodents have access to the poison bait (Broome and Garden 2013; Springer 2016). Such operations, pioneered on New Zealand's offshore islands, have a good track record in recent years with all 22 operations targeting mice being successful since 2005 (although seven of these islands were subsequently reinvaded; DIISE 2018 - http://diise.islandconservation.org data accessed 26/01/2018, Filter: House Mouse|Toxicant, sort by '*aerial baiting'* of toxic bait as the primary baiting method for years 2005-2017).

Gough Island, which is part of the United Kingdom's Overseas Territory of Tristan da Cunha, is a UNESCO Natural World Heritage Site and is regarded to be one of the most important seabird breeding islands in the world (Angel and Cooper 2006). The United Kingdom's Royal Society for the Protection of Birds is planning an eradication attempt on Gough Island in the winter of 2019. At 65 km^2, Gough will be the largest island where an eradication has been attempted targeting mice alone (Springer 2016). Planning for the Gough Island eradication has involved more than a decade of research to ensure the highest probability of success (e.g. Angel and Cooper 2006; Brown 2007; Parkes 2008; Wanless *et al*. 2009; Cuthbert *et al*. 2011, 2014, 2016).

The South African Department of Environmental Affairs is planning to mount an eradication attempt on Marion Island in the austral winter of 2020 or 2021. The Prince Edward Islands are South Africa's only oceanic islands and are recognized as Special Nature Reserves, affording them the highest level of protection under South Africa's National Environmental Management Protected Areas Act of 2003.

The planned eradication operation on Marion Island will be larger than any previous island eradication *targeting mice only* (cf. Springer 2016; Martin and Richardson 2017). At 293 km^2, Marion Island is almost five times larger than Gough Island, but the terrain is less rugged, and the presence of a largely un-vegetated interior above 800 m with few if any mice in winter makes an eradication attempt at Marion less challenging in some regards (Parkes 2016).

The timing of the eradication and non-target species

The intention is to commence the eradication operations during early winter, when mouse numbers are falling due to lack of food and cold conditions, increasing the likelihood of all animals consuming bait (see Parkes 2017, for further details on the crucial decision of 'when to bait' on Marion Island). Mice cease breeding from late May to August, reducing the chances of semi-independent young in the den failing to encounter bait (Parkes 2016). Winter also coincides with the period of lowest numbers of brown skuas and giant petrels *Macronectes* spp. present on the island, which might be killed accidentally by either primary or secondary poisoning. Mitigation plans will be needed to reduce the impacts on resident scavenging species (Wanless *et al*. 2010).

Resident bird species will require special management. At this stage, the intention is to keep approximately 100 lesser sheathbills *Chionis minor* in captivity during Marion's eradication attempt, given the moderate level of mortality of snowy sheathbills *C. albus* during the rodent eradication at South Georgia (Martin and Richardson 2017). The Prince Edward Islands are home to an endemic subspecies of sheathbill *C. m. marionensis*, with a total population of ~5,000 individuals. Fortunately, Prince Edward Island houses ~1200 birds of this subspecies and could be used to re-establish birds on Marion Island. Kelp gulls *Larus dominicanus* also are resident scavengers at Marion Island, but they may be less susceptible to non-target poisoning (Martin and Richardson 2017). Given the small population size (Table 7.2) and difficulty of catching and maintaining captive birds, there is currently no plan to mitigate impacts on this species. Gulls are thought to move freely between Marion and Prince Edward Island, so immigration should aid the recovery of the Marion population after the eradication. On Gough Island, the intention is to keep approximately 60 pairs of Gough buntings *Rowettia goughensis* and approximately 100 pairs of Gough moorhens *Gallinula comeri* for up three months after the operation.

Biosecurity

Eradicating rodents from islands is an effective, long-term conservation management action, provided robust biosecurity measures are put in place to minimise the likelihood of any reintroductions. The South African National Antarctic Programme has imposed stringent quarantine measures on all vessels and materials going to the Prince Edward Islands and Gough Island since the early 1990s (de Villiers and Cooper 2008; Cooper 2008; Prince Edward Islands Management Plan Working Group 1996). These include fumigation of the resupply vessel prior to each voyage, use of rat guards on all hawsers when in harbour, placement of rodenticide baits at strategic points throughout the ship, and inspection of all cargo before being opened ashore. However, these strict Biosecurity measures require dedicated personnel to ensure standards are maintained, since the best way to deal with invasive species is to prevent invasion in the first place.

Suggestions for future research and conclusion

Planning for mouse eradications at Gough Island has involved more than a decade of research to ensure the highest probability of success (Angel and Cooper 2006; Brown 2007; Parkes 2008; Wanless *et al*. 2009; Cuthbert *et al*. 2011a, 2011b, 2014, 2016; Dawson *et al.* 2015; Rexer-Huber and Parker 2011; Bond *et al.* 2016; Dagleish *et al.* 2017). Less focussed research has been conducted at Marion Island (Wanless *et al.* 2010; McClelland *et al.* 2018; Parkes 2017; Parkes 2016), but ongoing yearly monitoring of the impacts is important and this is usually carried out by the over-wintering biologists stationed on the islands – provided these research programmes continue. On Gough Island, for example, the recent observations of 'scalpings' on summer breeding albatross chicks was another sign that predation impacts are diversifying. On Marion, the frequency and spread of mouse injured albatross chicks is monitored by over-wintering biologists each year. This involves multiple week-long walks around the island to carefully scan cliffs and breeding sites for scalped or injured chicks. With the planned eradication still a few years away, this ongoing fieldwork is important to assess the yearly and overall impact.

Obtaining data on mouse home ranges would be interesting to explore how attacks on albatross chicks occur at scattered locations all around Marion Island's ~72 km coastline. Since there are generally multiple attacks at each affected site, it suggests some cultural transmission of this novel foraging technique, but how attacks were initiated seemingly independently at sites separated by distances hundreds of times greater than mouse home ranges is unknown. Individual mice could be tracked to gain some insight into their home ranges, similar to radio tracking work done on at mice Gough Island (Cuthbert *et al*. 2016) and rats on Christmas Island (Low *et al*. 2013).

On Marion, very few data exist on burrow-nesting petrel recruitment rates (i.e. fledglings returning to breed). This is largely because burrow-nesting petrels are difficult to work with and even if fledglings are banded with metal rings, their resights are very infrequent compared to surface nesting species. Although challenging, initiating a long-term programme to gather data would be very useful, especially after a successful eradication when on-island threats will be reduced and the at-sea threats will likely be the major drivers of population changes.

In conclusion, mice threaten seabird populations on both Marion and Gough Islands, but eradication is possible, given the successful eradication of invasive rodents from 516 islands (DIISE, 2018; Filter: Rodent|Successful|Toxicant). Once rodents are removed, the recovery of native species of plants and animals on islands can be spectacular (Bellingham *et al*. 2010; Jones *et al*. 2016). For example, only four years after the 2014 eradication of mice and black rats on Macquarie Island, blue petrels and grey petrels are breeding on the island (Springer 2016; K. Springer and J. Bird, pers. comm. Macquarie Island 2018). Another recent success story is Antipodes Island, which was declared mouse-free in March 2018 (see www.milliondollarmouse.org.nz). This is a particularly encouraging eradication success, since Antipodes Island has many similarities to Marion and Gough Islands (e.g. relatively large at 20 km^2 with a similar landscape where mice are the sole invasive mammal), and so provides a hopeful and recent benchmark of confidence to justify support for the Gough and Marion island plans.

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Appendix 1

The distribution and abundance of blue petrels *Halobaena caerulea* breeding at sub-Antarctic Marion Island

A blue petrel *Halobaena caerulea* at Swartkop, Marion Island 2012 (photo Delia Davies)

This appendix is broadly based on this publication:

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Author contributions: PGR & BJD planned and carried out the main field survey where all authors participated; BJD completed post-survey fieldwork, analysed data and wrote the draft; PGR analysed data and edited drafts; All co-authors contributed to drafts.

Appendix 1: The distribution and abundance of blue petrels *Halobaena caerulea* breeding at sub-Antarctic Marion Island

Abstract

Blue petrels *Halobaena caerulea* are known to breed at seven locations in the Southern Ocean. Population estimates recently have been made for the two major breeding sites, but accurate estimates are lacking for the remaining locations. We used a systematic survey technique to estimate the size of the population breeding at Marion Island (293 km^2), the larger of the two Prince Edward Islands. A combination of colony area and density estimates suggested there were 214,700 blue petrel burrows on Marion Island in 2012. Burrow occupancy rates at the mid-incubation stage averaged 82% (range 36–98%), suggesting a total breeding population of 145,000 pairs (95% confidence interval 110,000–180,000). There appeared to be some range expansion since the population was mapped in the mid-1980s. Predation of chicks and eggs by introduced house mice *Mus musculus* could be affecting the recovery of blue petrels since feral cats *Felis catus* were eradicated in 1991. Based on this count from Marion Island alone, the Prince Edward Islands support the third largest population of blue petrels globally, after Diego Ramirez Islands and the Kerguelen Islands.

Introduction

With populations of some species numbering in the tens of millions, burrowing petrels are the most abundant seabirds in the Southern Ocean (Brooke 2004a). They form an important component of regional ecosystems, especially on their breeding islands where they are key drivers of ecological processes such as marine nutrient imports, soil disturbance and vegetation dynamics (Dean *et al*. 1994; Smith *et al*. 2011). These processes, which are integral to maintain island biodiversity, are readily disrupted by the introduction of mammalian predators that can have catastrophic impacts on burrowing petrel populations (Croxall *et al*. 2012). Reliable population estimates are important for the long-term monitoring and conservation of burrowing petrels. Such estimates are particularly important to track the recovery of petrel populations on islands where introduced predators have been removed.

The blue petrel *Halobaena caerulea* is a small, burrowing petrel (≈220 g) closely allied to the prions and the only member of its genus. It breeds in dense and usually discrete colonies (Marchant and Higgins 1990) at six sub-Antarctic island groups: Diego Ramirez and other islands off southern Chile, the Kerguelen Islands, the Prince Edwards, the Crozets, South Georgia and Macquarie Island (Warham 1990; Marchant and Higgins 1990), and a small population was recently discovered breeding farther north at cool-temperate Gough Island (Ryan *et al.* 2015). It is classified as Least Concern by the IUCN (2015), with an estimated global population of ≈3 million adult birds (Brooke

2004a), but this is based on crude estimates of the populations at most breeding localities. More robust population estimates have been made at the two major breeding sites in the last decade: Diego Ramirez, with ≈1.35 million pairs, is estimated to support more than half the global population (Lawton *et al.* 2006); and Mayes Island in the Kerguelen Islands, has ≈142,000 pairs (Barbraud and Delord 2006). However, for the remaining breeding sites there are mostly crude, order of magnitude estimates, owing to the difficulty of counting nocturnal petrels which breed in burrows (Brooke 2004a). The sub-Antarctic Prince Edward Islands (46°54'S, 37°45'E) in the southwest Indian Ocean comprise two islands, both of which support populations of blue petrels: Marion (293 km^2) and Prince Edward (44 km^2). The distribution of breeding colonies on Marion Island was mapped in the 1980s (Fugler *et al*. 1987), but there are no accurate population estimates (Ryan and Bester 2008).

The major threat to burrowing petrels breeding on oceanic islands is the introduction of mammalian predators (Croxall *et al.* 2012). House mice *Mus musculus* were brought to Marion Island during the sealing era sometime before 1818 and in 1949 domestic cats *Felis catus* were introduced to control mice at the newly-established weather station (van Aarde 1980). The cats soon turned feral, greatly reducing burrowing petrel populations over four decades (Schramm 1986), before finally being eradicated by 1991 (Bester *et al.* 2002). In the absence of competition and predation from larger introduced species, and aided by a drier, warmer climate year-round, mice now attain very high population densities in summer (up to 237 mice·ha^{-1}, McClelland 2013).

My primary aim of this study was to estimate the size and distribution of the blue petrel population at Marion Island and to establish whether the island is globally important as a breeding site for the species. My secondary aim was to develop a practical method for comparable surveys in future. The census reported here provides a baseline against which future changes in the population can be judged. Burrowing petrel numbers on Marion Island were predicted to increase following the eradication of cats (Hunter 1990), but there are concerns that predation on eggs and chicks by mice is slowing the recovery of at least some petrels (Cerfonteyn and Ryan 2015; Chapter 3).

Methods and study area

Locating colonies

Historical studies on Marion Island reported that blue petrels bred on the coastal lowlands below 500 m (van Zinderen Bakker 1971; Schramm 1986). Fugler *et al.* (1987) mapped the rough distribution of blue petrel colonies on the island based on field observations from October 1981 to May 1983; however, coverage was only 'reasonably complete' (p 1) and few details are reported as the paper focuses on breeding biology. In 2012, my co-fieldworkers and I conducted a thorough survey from 18 April – 6 May of these areas and new areas where blue petrel colonies were known or thought to occur. Although this is outside of the petrel's breeding season (birds return to the island from early September and breed from mid-October to early February; Fugler *et al*. 1987), blue petrels were present as they return to the island (after a post-breeding moult period at sea from mid-April to mid-May) to re-occupy and renovate their burrows (Fugler *et al*. 1987; Cherel *et al*. 2016). This is also the only time when there is a sufficiently large team on the island to undertake such a labour intensive task. Some areas were not visited due to time constraints during the April-May 2012 relief visit and I counted these areas (108 of the 292 colonies and sub-colonies identified) between 20 September and 5 November 2012 when the petrels had returned to breed. To estimate the area of the island occupied by blue petrels I used actual ground surface area (i.e. not projected) of altitudinal zones taken from Meiklejohn and Smith (2008).

On Marion Island the higher elevations (>650 m) are dominated by a barren polar desert biome which is unsuitable for blue petrel burrows. The lowland areas are dominated by a tundra biome with four habitat complexes (Gremmen and Smith 2008): mire (wet and relatively flat boggy areas dominated by mosses *Sphagnum* spp. and the grass *Agrostis magellanica*); slope (areas with well-drained soils dominated by the creeping stems of *Blechnum penna-marina* ferns and or *Acaena magellanica* creeper which form large soft mats of vegetation); salt spray (coastal slopes and flat areas dominated by extensive low herbfields of *Cotula plumosa* and/or *Crassula moschata* and in some areas also large sprawling cushion plants *Azorella selago*); and biotic habitats (areas fertilised by seal and seabird colonies dominated by tussock grass *Poa cookii*, tufts of the sedge *Uncinia compacta*, and introduced grasses *Poa annua* and *Agrostis stolonifera*). I classified the areas where blue petrels bred into these broad vegetation categories: *Cotula* flats (including a few *Azorella* areas), *Poa cookii* tussock slopes and *Blechnum*/*Acaena.*

Blue petrel colonies were located using one or a combination of these indicators:

(1) A grouping of fairly small burrows, where the adults will often call when their calls are imitated or even when a person walks over their burrows (Crawford 1952; Fugler *et al.* 1987).

(2) Blue petrel remains from birds killed by sub-Antarctic skuas *Stercorarius antarcticus lonnbergi* (Adams 1982; Schramm 1983; Ryan *et al.* 2009b; Cerfonteyn and Ryan 2015). Skuas often remove the head and body of their smaller prey, so I relied on wing length (>220 mm) or the distinctive white-tipped tail feathers of blue petrels to differentiate from Salvin's prion (*Pachyptila salvini*) remains (wing <205 mm, Cerfonteyn and Ryan 2015).

(3) Patches of tussock grass, caused by the combined manuring effect of a concentration of birds (Smith 1976) were usually a good indication of a blue petrel colony (Schramm 1986). Salvin's prions also sometimes breed in small discrete clusters; however, I did not locate any within blue petrel colonies. Salvin's prion burrows are readily distinguishable as the size of their burrow entrances are slightly smaller (average blue petrel burrow entrance = 142 ± 23 mm wide by 110 ± 10 mm high, whereas prions are 112 ± 10 mm by 96 ± 9 mm; Chapter 3). When disturbed in their burrows, prions also readily respond with harsh chattering calls that are readily differentiated from blue petrel calls.

Estimating the number of burrows

I recorded burrow counts by area (e.g. Green Hill) and divided each area into different sites (e.g. Green Hill coastal slopes, Green Hill north-east slopes) which each supported multiple clusters of burrows. At each site I recorded altitude, aspect of slope, angle of slope, activity (calling, inactive) and percentage cover of the main vegetation types (see above).

I used a systematic survey technique at all known blue petrel colonies. Firstly, the surface area (estimated ‘colony area’ rather than ‘true surface area’) of each sub-colony was estimated either by pacing out the width and breadth (small sites, <20 m across and steep slopes, e.g. Puisie) or by walking the perimeter to calculate the area using a Garmin GPS (larger sites with gentle slopes >20 m across). The large areas where a GPS was used (planar area) were relatively flat (e.g. Swartkop plains) thus no correction was made for slope.

Secondly, the density of burrows at every site was estimated visually and assigned a density class (low, medium, high or very high). Thirdly, I used circular plots at a sample of colonies (n = 673 plots, range 4–71 plots per colony, see Table A1.4) to calculate the density of burrows in different habitat types around the island (Fig. 8.1). I counted all burrows (*n*) within circular plots of 1 m or 2 m radius (r), depending on the visually estimated density (2 m–radius plots for low and medium densities, and 1 m–radius for high and very high densities). Circular plots were placed randomly in each colony by throwing a walking stick and placing the centre of a plot where the pointed end of the stick landed. I calculated burrow density ($n/\pi r^2$) for each circular plot and used these to calculate the mean (± 95% confidence interval, CI) burrow density for each site assuming density estimates were normally distributed within each density class (Table A1.4). All burrows within a circular plot were counted and the proportion of old burrows noted (those burrows which looked overgrown or showed little fresh signs of use).

The number of burrows at each site was calculated as: (mean burrow$\cdot$m^{-2} for the site) × (surface area m^2). Colonies where burrow densities were not estimated (using circular plots) had a burrow density allocated based on the density class and habitat type. Small clusters of burrows (<100) were simply counted in groups of five to the nearest 5–10 burrows. Sites where I only found unoccupied burrows (no response to calls in April2012) showing little sign of recent activity, were revisited during the pre-breeding and incubation period (see above) to confirm whether birds were present.

Burrow occupancy and response probability

I checked the proportion of occupied burrows with once-off occupancy checks during the early-mid incubation period (3–24 November) at 30 trial sites around the island (numbered 1–30, Fig. A1.1) chosen to represent all habitat types and all four burrow density classes. At each trial site I selected a defined patch of burrows (average 35 ± 5 burrows, range 27–48). I recorded the minimum occupancy by recording the proportion of burrows where birds responded by calling when I imitated a blue petrel call at each burrow entrance, including old burrows (see Berrow *et al.* 2000).

Although blue petrels were very responsive, a small proportion of blue petrels present in burrows do not respond, so I calculated a response probability to correct the occupancy estimates. I tested the response to calls of incubating birds at marked study burrows that were fitted with observation hatches to view the nest chamber. Response rates were estimated for 47–48 known occupied burrows at each of three study sites (Fig. A1.1) representing the main blue petrel habitats: *Cotula* flats (Swartkops), *Poa cookii* tussock slopes (Blue Petrel Bay) and *Blechnum*/*Acaena* (Macaroni Bay). During 6–25 November 2012 I conducted occupancy checks at each site using the same call response protocol as in the occupancy trials. The mean response probability for each study site was the (number of birds responding) / (number of birds present).

I used two approaches to combine the estimates of burrow occupancy (from 30 trial sites) and response probability (from 3 study colonies). First, I used the response probability from all 3 study sites to correct occupancy estimates at the 30 occupancy sites irrespective of habitat differences. Second, I used the response probability from each habitat type to correct occupancy estimates based on habitat: *Cotula*, *Poa* and *Acaena/Blechnum*. The only habitat where response rate was not tested directly was *Azorella* cushions. This is a relatively minor habitat type for blue petrels on Marion, and was considered most similar in terms of burrow exposure to *Cotula* (at least during the incubation period), so I used the response probability from the *Cotula* study site for burrows in *Azorella*.

I estimated the proportion of occupied burrows at each of the 30 occupancy trial sites as:

$$\mathrm{O} = \frac{\mathrm{c/s}}{\mathrm{n}}$$

where O = the proportion of occupied burrows, c = number of burrows where a bird responded to call, s = response probability (by habitat type or averaged across all 3 sites), and n = total number of burrows in the occupancy site.

Estimating the number of breeding pairs

The 30 burrow occupancy trial percentages were applied directly to the colonies where occupancy was estimated. Those colonies where occupancy was not estimated had an occupancy estimate allocated based on an occupancy trial site which had a similar habitat type, location and density class. The number of breeding pairs (± 95% CI) at each colony was estimated as: (number of burrows) × (allocated or actual occupancy). These colony estimates were summed to provide the whole island estimate.

Data analyses were conducted in R version 3.1.2 (R Core Team 2014). I examined differences in response rates across the three habitat types with a Pearson's chi-squared goodness of fit test. I used Kruskal–Wallis chi-squared and post-hoc Tukey tests to determine the relationship between burrow density classes and occupancy rates. Burrow densities are reported as mean ± SD, statistical significance is set at $P < 0.05$ and 95% confidence limits for binomial proportions.

Results

Blue petrels occurred patchily on the coastal lowlands around Marion Island in areas with sufficient soil for burrowing; they were absent from recent black lava flows and from exposed grey lava ridges, which accounts for many of the coastal regions lacking colonies (Fig. A1.1). A total of 292 colonies and sub-colonies were recorded, which could be loosely aggregated into 14 local populations (Fig. A1.1). Blue petrels preferred gentle to moderate slopes (average 21 ± 11°, $n = 292$) such as the coastal *Cotula* herbfields around Swartkop; the steepest slope where breeding occurred was on the coastal *Poa cookii* slopes at Macaroni Bay (60°) and Puisie (50°). The average altitude of all colonies was 79 ± 46 m (range 13–220 m, $n = 292$) with the highest colony being found on the slopes of Repetto’s Hill in *Blechnum/Acaena* habitat. Colonies were found up to 2.5 km inland, with the farthest inland colony located on the lower southeast slopes of Mesrug (Fig. A1.1). The total area occupied by blue petrels was estimated to be 76 ha; equivalent to only 0.3% of the island area, and 0.7% of the area below 200 m.

Average burrow density for all sample plots was 0.48 ± 0.59 burrows·m^{-2} (95% CI 0.44–0.53 burrows·m^{-2}, $n = 673$ plots), with a maximum density of 4.46 burrows·m^{-2} in *Poa cookii* tussock on Puisie’s steep cliff slopes (Fig. A1.1). Colonies among *Poa* tussock had the highest average burrow densities (e.g. Puisie coastal cliffs and Blue Petrel Bay slopes, Table A1.1) with the lowest average burrow densities in *Azorella* habitat (e.g. La Grange). Multiplying burrow density (± SD) and surface area of each site gave an estimated 214,700 burrows on Marion Island (rounded to the nearest 100 burrows; 95% CI 168,300–261,300; Table A1.2).

Table A1.1. Average burrow densities (mean burrow·m^{-2}) of blue petrels in six vegetation types on Marion Island based on burrow counts in 673 circular plots.

Vegetation Type	Mean	SD	Max	Median	*n* plots
Acaena	0.381	0.435	2.546	0.239	190
Azorella	0.087	0.104	0.477	0.080	44
Blechnum	0.258	0.332	1.592	0.159	115
Cotula	0.427	0.388	1.910	0.398	176
Poa spp.	1.041	0.864	4.456	0.955	136
Agrostis	0.265	0.193	0.637	0.239	12

Once-off occupancy checks during the early-mid incubation period found 78 ± 11% of burrows showed signs of recently activity ($n = 1037$ burrows at 30 sites, range 47–90%) and the remaining burrows looked old and unused. Colonies north of Swartkop had the highest proportion of old burrows (average 53% in *Acaena* and 50% in *Cotula*, Table A1.3), and colonies in *Acaena* habitat at Green Hill, Puisie, Rook's and Storm Petrel Bay had the lowest proportion of old burrows (<15%, Table A1.4). Sites with very high burrow densities had the greatest proportion of old burrows (28%).

Birds responded to calls at 59 ± 13% (24–71%) of burrows in the once-off occupancy checks. There was a significant difference in the minimum occupancy rate (i.e. before correction for unresponsive birds) between the four burrow density classes (n = 1,037 burrows at 30 sites, $\chi^2_3 = 235.8$, $P < 0.001$, low 84 ± 6%, medium 77 ± 10%, high 80 ± 6%, very high 68 ± 2%), with sites classed as 'low' burrow densities having the highest average occupancy. Multiple comparisons of the means showed a significant difference between all classes ($P < 0.001$), except between classes low and high ($P = 0.165$).

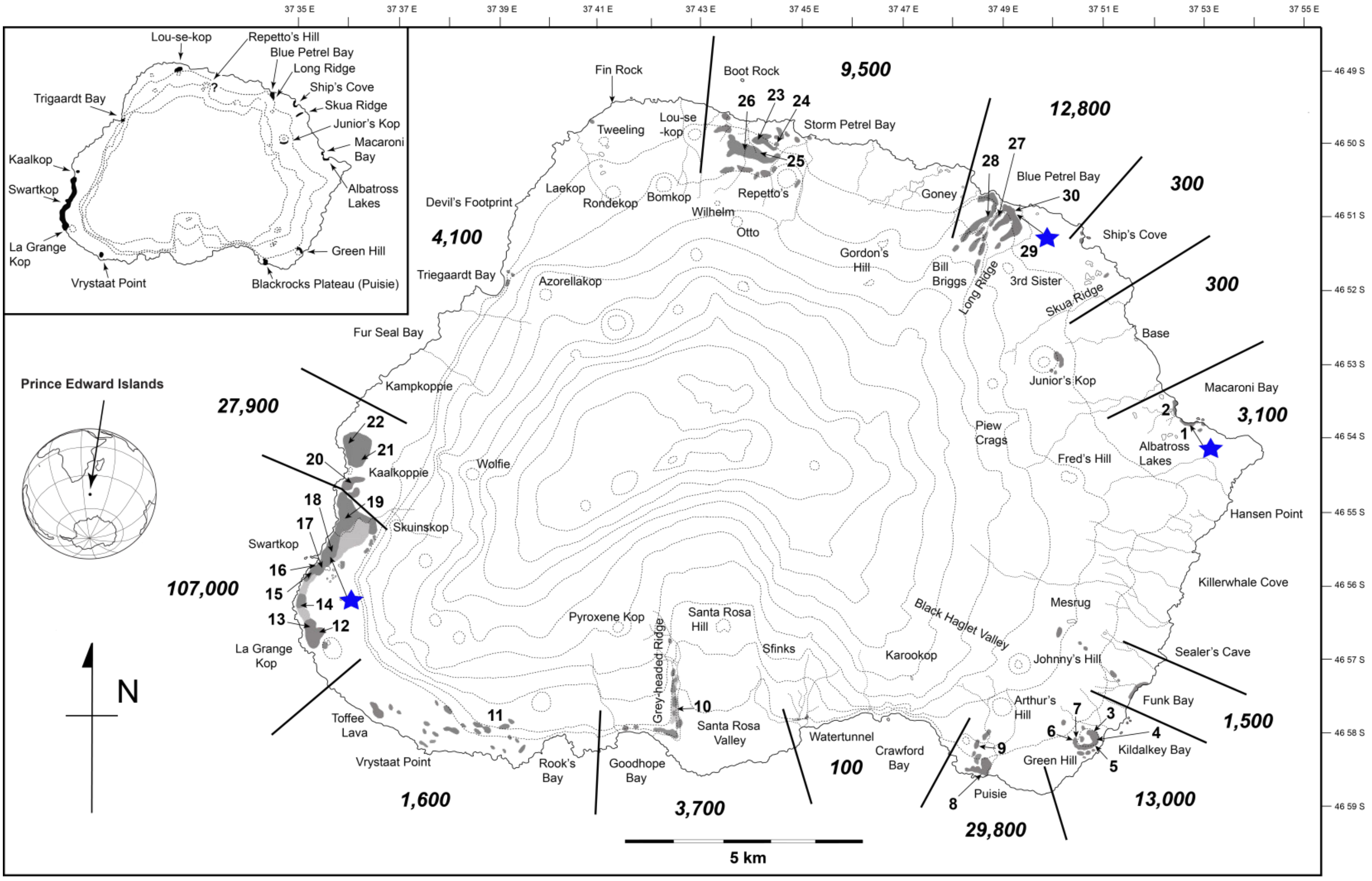

Figure A1.1. Distribution of blue petrel colonies at Marion Island (light shade = low and medium density, dark shade = high and very high density). Numbered sites are where burrow occupancy checks were conducted. Blue stars indicate study sites. Solid lines perpendicular to the coast loosely divide the 14 local populations into zones, and italic numbers are the estimates (before occupancy correction) of the number of blue petrel burrows (per zone rounded to the nearest 100). The inset map shows the historic range of blue petrel colonies from Fugler *et al.* (1987).

Table A1.2. The estimated number of blue petrel burrows and the number of breeding pairs (occupied burrows calculated using habitat-specific burrow occupancy) per region at Marion Island.

Area	**Burrows**	**± 95% CI**	**Breeding pairs**	**± 95% CI**
Junior's Kop	265	159–372	257	154–360
Macaroni Bay point and lake	964	709–1,219	699	508–890
Albatross Lakes	2,137	1,157–3,117	1,972	1,068–2,875
Kildalkey coastal slopes	319	221–417	276	191–360
Mesrug to coast at Funk Bay	1,142	868–1,417	987	750–1,224
Green Hill north-north-west slope (to Johnny's)	1,668	1,300–2,035	1,441	1,123–1,759
Green Hill north-east slopes	4,010	3,125–4,894	3,576	2,787–4,365
Green Hill south-east slopes	1,941	1,414–2,468	1,714	1,250–2,177
Green Hill south-west slopes	1,394	1,086–1,702	1,204	939–1,470
Green Hill west-north-west slopes	1,536	1,197–1,875	1,327	1,034–1,620
Green Hill Summit	2,248	1,752–2,744	1,943	1,514–2,371
Vegetated lava inland of Green Hill	232	167–297	201	144–257
Vegetated lava south of Green Hill	60	40–90	59	39–88
North-east of Puisie	1,025	427–1,622	861	363–1,358
Puisie scarp eastern face	11,453	7,091–15,816	10,239	6,339–14,139
Puisie main bay east side	3,193	2,612–3,774	3,090	2,528–3,653
Puisie main bay second level	8,259	6,756–9,761	7,317	5,986–8,648
Puise cliff top poa slopes	5,873	4,804–6,941	5,203	4,256–6,150
Watertunnel above hut and above gazella plain	130	110–150	112	95–130
Patches on Grey-headed Ridge inland of hut river	623	449–796	602	435–770
Patches on Grey-headed Ridge coastal of hut river	2,520	1,507–3,532	2,251	1,347–3,155
Slopes above Goodhope Bay	600	470–750	529	414–661
Coastal patches west of Rook's	315	245–375	275	214–327
Inland patches west of Rook's	480	385–565	415	333–488
Patches around Vrystaat Point	740	590–890	678	543–813
Toffee lava	150	120–190	130	104–164
La Grange north-west crater slopes	8,111	5,391–10,830	7,163	4,807–9,519
Cotula slopes north-west of La Grange	6,734	5,832–7,636	2,444	2,108–2,779
Coastal area from La Grange to Swartkop crator	3,507	2,496–4,558	1,472	1,048–1,912
Inland area from La Grange to Swartkop crator	713	498–929	290	203–378
Swartkop south crater slopes	18,422	15,681–21,164	6,579	5,600–7,558
Swartkop north crater slopes	5,538	4,650–6,425	1,978	1,661–2,295
Koppies around Swartkop hut	10,513	9,281–11,750	4,274	3,773–4,777
Patches inland from Swartkop	3,914	2,895–4,928	3,545	2,622–4,463
Coastal plains from Swartkop to Kaalkoppie	44,330	39,160–49,500	18,999	16,783–21,214
Cotula patches in black lava south-west of Kaalkoppie	1,910	704–3,115	819	302–1,335
Cotula plains south-west of Kaalkoppie	3,148	2,745–3,551	1,346	1,175–1,518
South of Kaalkoppie along recent lava hummocks/flats	4,210	2,973–5,447	3,718	2,626–4,811
North of Kaalkoppie inland from pathway cat trap	23,735	17,944–29,525	20,962	15,848–26,076
Triegaardt Bay north	4,021	3,025–5,017	3,569	2,685–4,453
West of Fin Rock (coastal *cotula* patch)	100	80–120	75	60–90
Cape Davis to Repettos	9,448	5,893–13,017	8,176	5,078–11,290
Sea Elephant Bay cliffs to Blue Petrel Bay	3,481	2,718–4,251	3,415	2,667–4,170
Blue Petrel Bay inland coastal slopes	486	291–681	477	286–669
Blue Petrel Bay patches along west tributary	354	212–496	347	208–486
Blue Petrel Bay patches along east tributary	575	344–805	564	338–790
Cliffs east of Blue Petrel Bay	1,194	1,030–1,357	1,171	1,010–1,332
Cliffs east of Blue Petrel Bay inland towards Bill Briggs	6,744	5,452–8,035	6,616	5,348–7,883
Ship's Cove	270	205–325	237	180–286
Skua Ridge, coastal end above King Bird head	10	5–15	9	4–13
Total Island count	**214,743**	**168,270–261,287**	**145,600**	**110,877–180,370**

Average response rate to calls from known occupied burrows at the three study sites was 72 ± 7% (*n* = 143 nests, range 67–77%), confirming that some birds do not respond, with no marked bias between the three habitat types ($\chi^2_2 = 2.002$, $P = 0.367$, *Cotula* 69%; *Poa* 76%; *Blechnum/Acaena* 75%). The low response rate in *Cotula* habitat, which is the most open habitat (at least during the incubation period), might be because petrels in short, open burrows are less likely to respond to calls (e.g. Ryan *et al*. 2006). Using the overall *average* response rate from known occupied burrows at the three study sites to correct burrow occupancy for all 30 occupancy sites gives an average occupancy of 82 ± 18% (*n* = 1037 nests, 33–99%), suggesting a total breeding population of 145,100 pairs (95% CI 110,200–180,100) on Marion Island in 2012. By comparison, using the *habitat*-specific response rates from known occupied burrows at the three study sites to correct burrow occupancy for the 30 occupancy sites (by habitat type) gives an average occupancy of 82 ± 17% (*n* = 1037 nests, 36–98%, Table A1.4), suggesting a total breeding population of 145,600 pairs (95% CI 110,900–180,400; Table A1.2). The two population estimates are very close, but using the habitat-specific occupancy correction does give a more accurate indication of the distribution of breeding pairs around the island, especially at high density areas, for example at Swartkop the *Cotula* coastal flats and crater slopes had ~7% more birds and at Puisie the *Poa* scarp slopes had ~6% less birds. The greatest concentrations of birds were found in *Cotula plumosa* herbfields along the coastal spray-zones at Swartkop (26% of total island count of breeding pairs) and in the *Poa* tussock at Puisie (18%).

Combining these results with recent data from the Diego Ramirez Islands (off Cape Horn) and Mayes Island (Kerguelen Islands), blue petrels occupy around 1.6 million burrows each year (95% CI roughly 1.18–2.26 million pairs). If I include all population estimates from known sites, the total breeding population is likely more than 2.3 million pairs (Table A1.5). The Prince Edward Islands probably support the third largest population, after Diego Ramirez and Kerguelen.

Discussion

Historical records

The first records of blue petrels on Marion Island are from the early 1950s. Crawford (1952) described how 'their presence was only made known by the fortuitous discovery of a breeding colony through the birds betraying themselves by chirping loudly in their holes...on the side of a steep grassy cliff....a few hundred in number' (p. 79). Rand (1954) described how 'despite wide-spread digging....[blue petrels] were first encountered [below Junior Kop] in numbers in April when their continual vocal activity made their whereabouts unmistakeable' (p. 197), which gives the impression that breeding colonies of blue petrels were not very common around the Meteorological Base Station in 1951, since they would surely have encountered colonies elsewhere, or remains of depredated birds at skua middens during their 'regular [monthly] coastal surveys' (Rand 1954, p. 173). van Zinderen Bakker (1971) reported that blue petrel nests 'were found between sea level and an altitude of 500 meters' and that blue petrels 'show a distinct preference for dry slopes of fine scoria on which *Acaena adscendes* [*magellanica*] grows... and dry slopes covered with *Poa cookii*' (p. 167).

The first crude population estimates were made in the mid-1970s, when cat numbers peaked, by Williams *et al.* (1979), who estimated that 'tens of thousands' of pairs bred on Marion Island. In 1979–80, Schramm (1986) investigated nest site preferences and burrow densities of burrowing petrels on Marion Island and concluded there were ≈6,000 blue petrel burrows in his 1,040 ha study area in the northeast sector of the island, where they were mostly confined to steep vegetated slopes along the coast. I estimate ≈15,400 occupied burrows for this area, but my sampling approach is quite different from Schramm's (1986) random transects, so cannot be compared directly. I repeated Schramm's transects in 2012/13 (Chapter 3) and estimated there were 9,300 blue petrel burrows, but this is a very crude extrapolation given the low proportion of transects containing blue petrels in this area, and so I have little confidence in assessing any population change based on these data. Based on field experience, Ryan and Bester (2008) suggested that at least 100,000 adult blue petrels (50,000 pairs) bred at Marion Island. However, this study has provided the first quantitative estimate for the whole island, of 145,000 pairs.

Survey technique

This survey technique proved to be a practical census method for a highly aggregated burrowing petrel breeding on a relatively large island such as Marion Island (293 km^2) and is recommended for comparable surveys in future. Random transects would have been inappropriate given the very small proportion of the island occupied by blue petrels (<1% of the area below 200 m elevation). A comprehensive survey of all colonies on Marion Island was made possible by the often distinctive vegetation associated with dense petrel breeding areas, which made it fairly easy to locate colonies. The approach also was facilitated by the relatively flat terrain, which allowed access by foot to all areas where blue petrels may breed and relatively easy calculation of surface areas by pacing or using a GPS (GPS-measured planar areas tend to underestimate surface area in steep terrains). Some sections of the coastline are inaccessible by foot (≈5%), but these are steep cliffs of barren volcanic rock unsuitable for blue petrel burrows. This survey technique requires a team of 4–5 field workers for the initial round island counts to locate and sample all blue petrel colonies and 1–2 overwintering field workers to check for areas missed and to complete the occupancy trials. Although time consuming, I believe that this method provides a reasonable estimate of the breeding population. As with all surveys of this nature, the confidence intervals are wide as they are so dependent on a number of variable factors (response rate, breeding propensity, survey effort, timing of the survey regarding early egg failures and different field workers). The timing of the occupancy trials are especially important and in this regard my estimate is likely an underestimate since a proportion of nests likely failed before occupancy sampling took place at mid-incubation. Hence, the ability to determine small to moderate population changes in future surveys is probably quite low; however, large changes in numbers and distribution around the island could be detected.

The impact of cats

Cats were present on Marion Island from 1949–1991 and by the mid-1970s an estimated 2,000 cats were killing some 450,000 birds per year, most of which were burrowing petrels (van Aarde 1980). Cats were seen to enter burrows of Salvin's Prions with 'distinct cat trails observed running from burrow entrance to burrow entrance' (van Aarde 1980, p. 126). A study of cat prey items found Salvin's Prions to be by far the most common prey (60%, *n* = 1,224); blue petrels comprised only 2.5% of prey remains (van Aarde 1980). However, the small study area (≈43 ha) around the Meteorological Base Station did not include any large blue petrel colonies. Considering blue petrels have slightly larger burrows than Salvin's Prions, cats almost certainly would have been able to access their nest chambers and chicks and adults were likely the primary prey items for cats which had blue petrel colonies within their territories.

Fugler *et al.* (1987) investigated the breeding biology of blue petrels on Marion in the early 1980s and expressed concern that, in the absence of a cat control programme, the levels of cat predation could cause the extermination of the species as a regular breeder as had happened at Macquarie Island, where the few remaining blue petrels were restricted to cat-free offshore stacks (Brothers 1984). At the time, the cat control programme was in its early phases and it was estimated that there was a ≈70% increase in cat predation on blue petrels from 1975 to 1982 (van Rensburg 1985). Fortunately cats were eradicated, and blue petrel breeding success improved from 24% in 1982/83 (Fugler *et al.* 1987), to 64% in 1991/92 (Cooper *et al*. 1995) and 61% in 2012/13 (Chapter 5).

Changes in distribution since the 1980s

Blue petrels typically still occurred in the areas mapped by Fugler *et al.* (1987; inset map, Fig. A1.1). However, I found them in several new areas; most were small, isolated colonies (e.g. along the coastal slopes of Funk Bay and isolated colonies inland to Mesrug; at Watertunnel; and small colonies between Rook's Bay and Vrystaat Point, see Fig. A1.1). However, more extensive 'new' colonies were found on lowland slopes between Cape Davis and Repetto's (4.5% of the island count) and along Grey-headed Ridge (0.2%). It is not clear whether birds breeding in these areas were overlooked by Fugler *et al*. (1987), or whether there has been a range expansion at Marion Island since the 1980s. The latter hypothesis is supported since some of the colonies mapped by Fugler *et al*. (1987) appear to be more extensive now, e.g. at Blue Petrel Bay where colonies now extend up both valleys up to 140 m elevation, whereas previously colonies apparently were only along the coast below 100 m; and larger colonies (than indicated in Fugler *et al*. 1987) at Green Hill, Puisie and Kaalkoppie (in mixed *Acaena/Poa* habitat with 13% of the total island count). Three colonies appear to have either become smaller or disappeared entirely since the 1980s (Fugler *et al.* 1987): at Ship's Cove, Skua Ridge and Vrystaat Point. The colony at Ship's Cove occurred on steep coastal slopes that have been covered in a dense mat of the invasive grass *Agrostis stolonifera* (Gremmen *et al.* 1998), making them unsuitable for burrowing petrels.

Threats to the global population

Introduced predators pose the greatest threat to blue petrels. Cats and rats *Rattus* spp. extirpated blue petrels from Cochons and Possession (Crozet Islands, Jouventin *et al.* 1984) and from Macquarie Island, although small numbers persist on offshore stacks (Brothers 1984). Blue petrels are also 'very scarce' on Grande Terre (Kerguelen Islands) due to predation by cats and rats (Barbraud and Delord 2006). The impacts of rats on blue petrels at South Georgia are not well known, but probably have been significant. Clarke *et al*. (2012) report that blue petrels "breed widely on the south coast and offshore islands, where rats are absent" (p. 133), which supports the assertion by Poncet (2006) that the smaller petrels have been largely eliminated from rat-infested coastal areas of South Georgia.

The impacts of introduced house mice on blue petrels are less clear. Fugler *et al.* (1987) suspected that mice ate some eggs and chicks at Marion Island, but Mayes Island has a large population of blue petrels despite the presence of mice (Barbraud and Delord 2006). Wanless *et al*. (2007) hypothesized that the impacts of mice on seabirds are most severe where they are the only introduced mammal because their populations are not regulated by the effects of competition and predation by other, larger introduced species (e.g. cats or rats). At Marion Island, the removal of cats, combined with a warmer and drier climate (Le Roux and McGeoch. 2008), probably have allowed mouse densities to increase in summer (Ferreira *et al*. 2006). In 2008–11, densities in mire habitats reached up to 237 mice·ha^{-1} (McClelland 2013), similar to peak densities on Gough Island (266 mice·ha^{-1}, Cuthbert *et al*. 2016), where mouse predation severely affects burrowing petrel populations (Chapter 4). The recent report of widespread mouse attacks on large, well feathered albatross chicks at Marion is cause for concern (Chapter 6). Predation of chicks and eggs by introduced mice could explain the slow recovery of petrels over the last two decades since cats were eradicated. In the northeast of the island there has been only a marginal increase in burrow densities since 1979 (Schramm 1986; Chapter 3).

Currently, most large populations of blue petrels occur at islands lacking introduced predators: Diego Ramirez, Prince Edward Island, Est and Pingouins (Crozet Islands), and at least 18 predator-free islands in Golfe du Morbihan (Kerguelen) together support at least 80% of the global population (Table A1.5). It is impossible to estimate what the global population might have been before the spread of introduced mammals throughout the sub-Antarctic. Fortunately it is possible to restore breeding islands by eradicating invasive mammals (Clout and Veitch 2002). In April 2014, Macquarie Island was declared free of introduced mammals after a successful eradication programme in 2011, and seabirds are already returning to numerous areas of the main island (Rachael Alderman, pers. comm.). An eradication attempt has also been concluded at South Georgia; its success remains to be confirmed, but initial signs are positive (Neil 2016). Eradicating mice from Marion Island would benefit not only the burrowing petrel populations but also help to restore the original structure and functioning of the island's terrestrial ecosystems.

Table A1.3. Estimates (mean burrow·m^{-2} ± 95% confidence interval) of burrow densities (before occupancy correction) of blue petrels at selected sites on Marion Island based on burrow counts in 673 circular plots.

Area	***n* plots**	**Type**	**Density class**	**Estimate**	**95% lower**	**95% upper**
Junior's Kop	20	*Blechnum*	low	0.036	0.006	0.065
Macaroni Bay *Poa* slopes	4	*Poa*	very high	1.353	1.048	1.658
Macaroni Bay cliff tops/inland	11	*Blechnum*	low	0.188	0.062	0.314
Albatross Lakes	15	*Poa/Blechnum*	medium	0.562	0.305	0.820
Kildalkey	5	*Acaena*	low	0.271	0.207	0.334
Green Hill	67	*Acaena/Blechnum*	medium	0.749	0.584	0.915
Puisie cinder cone	7	*Acaena*	low	0.171	0.076	0.265
Puisie eastern scarp	11	*Acaena/Poa*	high	1.215	0.777	1.654
Puisie main bay	32	*Cotula/Poa*	very high	1.835	1.501	2.169
Grey-headed Ridge	20	*Acaena*	medium	0.279	0.166	0.391
La Grange	48	*Acaena*	medium	0.290	0.201	0.379
La Grange	20	*Azorella*	low	0.095	0.039	0.152
La Grange, Swartkop to Kaalkoppie	16	*Cotula* low	low	0.094	0.048	0.141
La Grange, Swartkop to Kaalkoppie	71	*Cotula* medium	medium	0.423	0.369	0.476
La Grange, Swartkop to Kaalkoppie	22	*Cotula* high	medium	0.619	0.543	0.694
Kaalkoppie south slopes	30	*Cotula*	low	0.048	0.018	0.078
Kaalkoppie (*Cotula* patches)	10	*Cotula*	medium	0.438	0.323	0.552
Kaalkoppie north	63	*Acaena*	low	0.162	0.122	0.201
Triegaardt Bay flats	5	*Cotula*	low	0.175	0.005	0.345
Triegaardt Bay cliffs	20	*Poa*	high	0.967	0.746	1.188
Repetto's to Cape Davis at Wilhelm River	9	*Acaena*	medium	0.460	0.246	0.673
North of Wilhelm River	42	*Blechnum*	low	0.157	0.099	0.215
Inland of Storm Petrel Bay cliffs	25	*Acaena/Blechnum*	medium	0.236	0.146	0.325
Amphitheatre above Storm Petrel Bay	6	*Acaena/Poa*	high	0.623	0.433	0.814
West of Wilhelm River, upper slopes	32	*Blechnum*	medium	0.458	0.070	0.229
Long Ridge West Point and Sealers' Beach	36	*Poa*	very high	1.194	1.033	1.354
Long Ridge Sea Elephant Bay slopes	26	*Poa*	low	0.195	0.022	0.367

Table A1.4. One-off burrow occupancy estimates of blue petrel burrows during the early incubation period at 30 trial sites around Marion Island. The proportion of occupied burrows (% occupied) was estimated by callback at each trial site and corrected using *habitat*-specific response probabilities from three study sites representing different habitat types.

Location (map references, see Fig. A1.1)	Type	Density class	*n* Burrows	% Occupied
Macaroni Bay (1)	*Poa*	medium	33	71
Macaroni Bay (2)	*Blechnum*	low	32	91
Green Hill north-north-east slopes (3)	*Blechnum*	very high	36	92
Green Hill north-east slopes (4)	*Acaena*	very high	40	97
Green Hill south-east coastal slopes (5)	*Poa*	very high	38	89
Green Hill west slopes (6)	*Blechnum*	medium	48	86
Green Hill interior north-west slopes (7)	*Acaena*	high	31	89
Puisie main bay slopes (8)	*Poa*	very high	41	89
Puisie scoria slopes (9)	*Acaena*	high	35	95
Grey-Headed Ridge (10)	*Blechnum*	low	30	97
Rook's to La Grange, lower escarpment slopes (11)	*Acaena*	high	30	97
La Grange (12)	*Acaena*	medium	29	91
La Grange (13)	*Azorella*	medium	27	67
La Grange (14)	*Cotula*	high	31	92
Swartkop south slopes towards tea gardens (15)	*Cotula*	high	38	91
Swartkop upper south slopes of amphitheatre (16)	*Cotula*	very high	32	80
Swartkop plains east of amphitheatre (17)	*Acaena*	very high	30	60
Swartkop plains north-east of amphitheatre (18)	*Acaena*	very high	34	41
Swartkop, over river towards Kaalkoppie (19)	*Cotula*	very high	42	43
Swartkop, further north towards Kaalkoppie (20)	*Cotula*	very high	42	36
Kaalkoppie, south-east slopes (21)	*Acaena*	medium	36	88
Kaalkoppie, south-west slopes (22)	*Cotula*	high	31	97
Storm Petrel Bay, before amphitheatre (23)	*Acaena*	medium	31	98
Storm Petrel Bay, north of amphitheatre (24)	*Poa*	high	31	88
Storm Petrel Bay, inland of amphitheatre (25)	*Blechnum*	low	32	82
River crossing at the base of Wilhem (26)	*Acaena*	high	38	84
Blue Petrel Bay east side up valley (27)	*Acaena*	low	32	86
Blue Petrel Bay west side up valley (28)	*Cotula*	very high	40	75
Long Ridge coastal (29)	*Poa*	very high	36	83
Long Ridge coastal (30)	*Blechnum*	low	31	98
Summary			1037	82 ± 17%

Table A1.5. Estimates of populations of blue petrels at all seven known breeding locations, listed in descending population size. Estimates rounded to the nearest 1,000 pairs.

Location (area km^2)	Breeding pairs (95% CI)	Sampling method	Year	Source	Introduced mammals
Diego Ramirez Islands (1.4)	1.35 million (1.15–1.54)	transect & line-distance sampling	2002	Lawton *et al.* 2006	none
Kerguelen Islands (7,200)					
18 Islands in Golfe du Morbihan (23)	400,000–800,000	extrapolated by area	2002	Barbraud and Delord 2006	some have rats/mice/rabbits
Mayes Island (2.7)	142,000 (106,000–179,000)	5 m circle plots & transects	2002	Barbraud and Delord 2006	mice
Grand Terre (6.7)	very scarce	estimate based on field experience	2002	Barbraud and Delord 2006	cats/mice/rats/rabbits
Prince Edward Islands (337)					
Prince Edward Island (44)	100,000	estimate based on field experience	2008	Ryan and Bester 2008	none
Marion Island (293)	145,000 (110,000–180,000)	survey & 1 m / 2 m circle plots	2012	This study	mice (cats before 1991 eradication)
Crozet Islands (352)					
Est (130)	10s of 1,000s	estimate based on field experience	1984	Jouventin *et al.* 1984	none
Pingouins (3)	10s of 1,000s	estimate based on field experience	1984	Jouventin *et al*. 1984	none
Apôtres (2)	1,000s	estimate based on field experience	1984	Jouventin *et al*. 1984	none
Possession (150)	probably extinct	no recent surveys	-	Delord pers. comm. 2016	rats
Cochons (67)	probably extinct	no recent surveys	-	Delord pers. comm. 2016	cats/mice/rabbits
South Georgia (3,479)					
South Georgia Island (3,528)	70,000	estimate based on field experience	1980	Prince and Croxall 1983	rats/mice (eradication underway)
Bird Island (5)	10,000+	estimate based on field experience	1980	Croxall and Prince 1980	none
Macquarie Island (128)	500–600	ground counts	1982	Brothers 1984	(rats/cats/mice <2014 eradication)
Gough Island (65)	200+	estimate based on field experience	2014	Ryan *et al.* 2015	mice
Global total	**>2,3 million pairs**				

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Appendix 2

The distribution and abundance of white-chinned petrels *Procellaria aequinoctialis* breeding at the sub-Antarctic Prince Edward Islands

A white-chinned petrel *Procellaria aequinoctialis* and its chick in a burrow on Marion Island, 2013 (photo Ben Dilley).

This appendix is broadly based on this publication:

Ryan, P.G., Dilley, B.J. and Jones, M.G.W., 2012. The distribution and abundance of white-chinned petrels (*Procellaria aequinoctialis*) breeding at the sub-Antarctic Prince Edward Islands. Polar Biology, 35(12), pp.1851–1859.

Author contributions: PGR, BJD & MGWJ planned and carried out the Marion survey; BJD completed the post-survey Marion fieldwork; PGR & MGWJ completed the Prince Edward fieldwork; PGR, BJD and MGWJ analysed the data and wrote the draft.

Appendix 2: The distribution and abundance of white-chinned petrels *Procellaria aequinoctialis* breeding at the sub-Antarctic Prince Edward Islands

Abstract

I estimated the size of the white-chinned petrel *Procellaria aequinoctialis* population breeding at the Prince Edward Islands using a systematic survey method. In Chapter 3, I report how white-chinned petrels on Marion Island showed the greatest increase in burrow densities since the cat-era (1979), which was surprising considering white-chinned petrels are listed as vulnerable to extinction due to incidental mortality on fishing gear at sea and are the seabird most often killed on longlines in the Southern Ocean. In this systematic survey, I found white-chinned petrel burrows are largely confined to deep, muddy soils, usually on slopes below 200 m elevation, but locally up to 420 m. After correcting for count bias, Marion Island has an estimated 29,900 nests (95% CI 27,700–32,400). Burrow occupancy rates at the start of the incubation period were 65% during one-off surveys, but repeat surveys found that at least 73% of burrows were occupied and 87% of burrows showed signs of occupancy. This suggests that there were roughly 24,000 occupied nests on Marion Island (95% CI 20,000–28,000). A more cursory survey on Prince Edward Island yielded 14,700 burrows, suggesting that there are 9,000–15,000 occupied nests. Globally, white-chinned petrels occupy approximately 974,200 nests (95% CI 678,000–1,286,000), with the Prince Edward Islands the third most important breeding site, after South Georgia and Kerguelen. This systematic survey of Marion Island burrows provides a baseline against which future population changes can be assessed, which is especially important in light of the future mouse eradication operation.

Introduction

The white-chinned petrel *Procellaria aequinoctialis* is the largest burrow-nesting petrel and breeds at several sub-Antarctic islands. Despite a population of several million birds (Brooke 2004), it is listed as vulnerable (BirdLife International 2010) because white-chinned petrels are the seabird most often killed on longlines in the Southern Ocean and adjacent temperate waters (Barnes *et al.* 1997; Nel *et al.* 2002; Delord *et al.* 2005; Robertson *et al.* 2006; Petersen *et al.* 2009a, b; Delord *et al.* 2010) and is also impacted by other fisheries (e.g., Waugh *et al.* 2008; Watkins *et al.* 2008). Breeding populations appear to have decreased at almost 2% per year since the early 1980s both at Bird Island, South Georgia (to 1998, Berrow *et al.* 2000) and Ile de la Possession, in the Crozet Islands (to 2004, Barbraud *et al.* 2008). The density at sea in Prydz Bay, east Antarctica, fell dramatically from 1980 to 1992 (Woehler 1996) and numbers following research ships in the southwest Indian Ocean decreased 35% from the 1980s to 2000s (Péron *et al.* 2010).

Compared to surface-nesting species such as albatrosses and giant petrels, the world population of white-chinned petrels is poorly known. To monitor the global population trends and assess the impacts of fishing mortality on the species, an estimate the world population is needed. White-chinned petrels breed in burrows and are largely nocturnal, making their breeding populations difficult to count accurately. Until recently, there were only crude, order of magnitude estimates of population sizes for almost all colonies (Brooke 2004). However, more accurate estimates recently have been made for four breeding sites: South Georgia (Martin *et al.* 2009), Kerguelen (Barbraud *et al.* 2008), the Crozets (Barbraud *et al.* 2009), and the Falklands/Malvinas (Reid *et al.* 2007). This leaves the Prince Edward Islands as the last breeding site of the nominate subspecies for which a population estimate is lacking. In this chapter, I estimate the numbers of white-chinned petrel burrows on the Prince Edward Islands using a systematic survey method and I estimate the size of the breeding population based on burrow occupancy estimates from Marion Island. In this study, I aim to (1) develop and complete a systematic survey of Marion and Prince Edwards Islands burrows to provide a baseline against which future population changes can be assessed, which is especially important in light of the future mouse eradication operation on Marion Island; and (2) to combine these counts with estimates for all other populations of the nominate subspecies to provide the first total population estimate for *P. a. aequinoctialis.*

Methods

Timing of the Marion Island systematic survey

White-chinned petrels breed in loosely clustered colonies (Brooke 2004), which made it feasible to locate all breeding locations and to count their burrows (Barbraud *et al.* 2008). I estimated numbers of white-chinned petrel burrows on Marion Island (293 km^2) during 2009. Most burrows were counted during a comprehensive survey from April 16 to May 5, 2009, which is the only time of year when there is a sufficiently large team on the island. It is the end of the breeding season for white-chinned petrels, which is not the ideal time to conduct the survey because there is little activity at most nests. However, the number of burrows does not change seasonally, and subsequent fieldwork during the pre-breeding and incubation period (October–December), when adults are highly vocal (Berrow 2000), confirmed that significant populations had not been overlooked.

Identifying and counting burrows

Most white-chinned petrel burrows are readily identified by their large size, often with an entrance pool or moat (van Zinderen Bakker 1971), and the presence of tussock grass *Poa cookii* at most colonies (Schramm 1986). I surveyed all suitable-looking habitat, paying particular attention to areas with tussock grass. The identity of burrows at high elevation, inland sites was confirmed by one of three cues: (1) adults arriving at dusk, (2) the presence of fledglings (alive or dead; many fledglings are killed by sub-Antarctic skuas *Catharacta antarctica*), or (3) by subsequent visits to sites at the start of the breeding season.

Counting and checking the status of each burrow entrance are not practical, given the large numbers of burrows. For example, to determine the number of nests, you would have to check that *each* burrow entrance leads to a nest chamber; potential biases include collapsed burrows, multiple entrances which all lead to a single nest chamber, and multiple nest chambers linked to a single burrow entrance. To deal with this, I crudely estimated numbers of burrow entrances to the nearest 5–10 burrows at each colony. The accuracy of these estimates was checked subsequently by carefully counting the number of nests at 24 sites located throughout the species' range around Marion Island, representing the full range of habitats used. At each check site, which contained 30–60 nests, burrow entrances were checked to ensure the nest had not collapsed, whether there was more than one entrance to a single nest or whether one burrow entrance led to more than one nest. I used the ratio of the estimated number of burrow entrances to the number of nests to estimate total nest numbers. This approach has been used successfully to track population changes in the closely related spectacled petrel *Procellaria conspicillata* at Inaccessible Island (Ryan and Moloney 2000; Ryan *et al.* 2006; Ryan and Ronconi 2011).

Assessing burrow occupancy

Rates of nest occupancy were checked during the early incubation period from 28 November to 14 December 2009, at 15 sites around Marion Island (Fig. A2.1a). Nest occupancy was assessed using three criteria:

(1) A tape of white-chinned petrel calls was played down the burrow entrance for up to 30 seconds, ceasing immediately once a response was obtained. The tape included both rattle and wheezy calls of both sexes, so should evoke a response from most incubating adults (Berrow 2000; Barbraud *et al.* 2008).

(2) The burrow entrance was examined for signs of recent activity: fresh vegetation (used as nest lining), fragments of egg shell (indicative of a failed nest), feathers, fresh guano, fresh digging or footprints, and freshly cropped vegetation. Burrows with large moats could not be checked for signs of occupancy.

(3) A flexible probe was inserted as far as could be reached (approximately 1.5 m) down burrows where there was no response to playback, to try to feel whether the nest was occupied, and the bird had not responded to the tape.

Five repeat checks of 100 marked burrows (50 at each of two sites) were made to test the accuracy of single nest occupancy checks (cf. Ryan *et al.* 2006). Repeat checks were made from 20 to 28 November 2009, close to the base on the northeast coast of Marion Island (Fig. A2.1a).

The Prince Edward Island survey

The distribution of white-chinned petrel burrows at neighbouring Prince Edward Island (49 km^2) was mapped during a brief visit from 16–22 December 2008, augmented during subsequent short visits in April 2010 and March 2011. Coverage was less comprehensive than that at Marion Island, due to the limited time available on the island, so the estimate is likely to be conservative. There was insufficient time to estimate burrow count accuracy and visits took place at the wrong time of year to estimate burrow occupancy rates; in the absence of data specific to Prince Edward Island, these estimates were assumed to be the same as on Marion Island, even though some different observers were involved in the Prince Edward Island surveys, potentially affecting count accuracy.

Density estimates in white-chinned petrel colonies on Marion Island were made in 36 m^2 circular plots, following the method used by Berrow *et al.* (2000). Sites were selected at random by throwing a marker within the confines of a colony. Our intention was to test whether burrow densities differed between the two Prince Edward islands, but there was insufficient time to conduct density plots on Prince Edward Island. We report the Marion data for comparison with data from South Georgia (Berrow *et al.* 2000; Martin *et al.* 2009).

Global population estimate

A total population estimate for white-chinned petrel was obtained by combining the breeding population at the Prince Edward Islands with estimates from all other breeding localities (Barbraud *et al.* 2008, 2009; Martin *et al.* 2009). This was converted into a total population of fully grown birds by assuming breeding adults represent roughly 44% of the total population, based on the demographic model for the Crozet population following Barbraud *et al.* (2008).

Results

Distribution of burrows

White-chinned petrels breed on gently sloping areas around most of the coastal plain of Marion Island (Fig. A2.1a), with gaps in their distribution where there are large black lava flows (e.g., Blackrocks Plateau, Santa Rosa Valley, Devil's Footprint) or sheer sea cliffs (e.g., above Crawford and Triegaardt Bays). However, along much of the southwestern coast, breeding is confined to the inland scarp rather than along the coastal margin, where soils are too shallow to support burrows (Fig. A2.1a). Populations extend locally up to 3.7 km inland at Hendrik Fister Kop and below Piew Crag in the northeast, and up to 420 m elevation between Bomkop and Spitskop on the north coast, and 330 m east of Pyroxene Kop on the south coast (Fig. A2.1a).

Burrow and population estimate

A total of 30,800 burrows was estimated at Marion Island, with the greatest concentrations along the northwest coast between Kaalkop and Triegaardt Bay (25% of all burrows), and between Cape Davis and Repetto's Hill (16%, Fig. A2.1a). Careful checking of burrows found that observers overestimated the number of nests (burrows/nests = 1.03 ± 0.04, range 0.80–1.23; n = 850 nests at 24 sites). Most of the errors resulted from either old, collapsed burrows, or nests with multiple entrances. There was no obvious difference in bias linked to habitat. Applying this correction factor suggests that there were 29,900 nests on Marion Island (95% CI 27,700–32,400). Burrow density in white-chinned petrel colonies averaged $0.166 \pm 0.065 \cdot m^{-2}$ (n = 100, range 1–13 per 36 m^2 plot). Of 665 white-chinned petrel nests checked, 34% had entrance pools or moats, but the proportion varied markedly between sites and tended to be greater at coastal sites (Table A2.1). One-off occupancy checks of these nests during the early incubation period found that 87% showed signs of recent use (see Methods for details) and at least 65% were occupied by petrels. However, multiple checks of 100 marked nests found that 73% were occupied on at least one of five visits (compared to a maximum of 65% occupancy on any single check). Accordingly, I assumed that 73–87% of nests were occupied, giving a breeding population in 2009/10 of 21,800–26,000 pairs (best estimate 24,000 occupied nests; 95% CI 20,000–28,000) on Marion Island. This is likely to be a conservative estimate, because some small colonies may have been overlooked and some cryptic nests would have been missed.

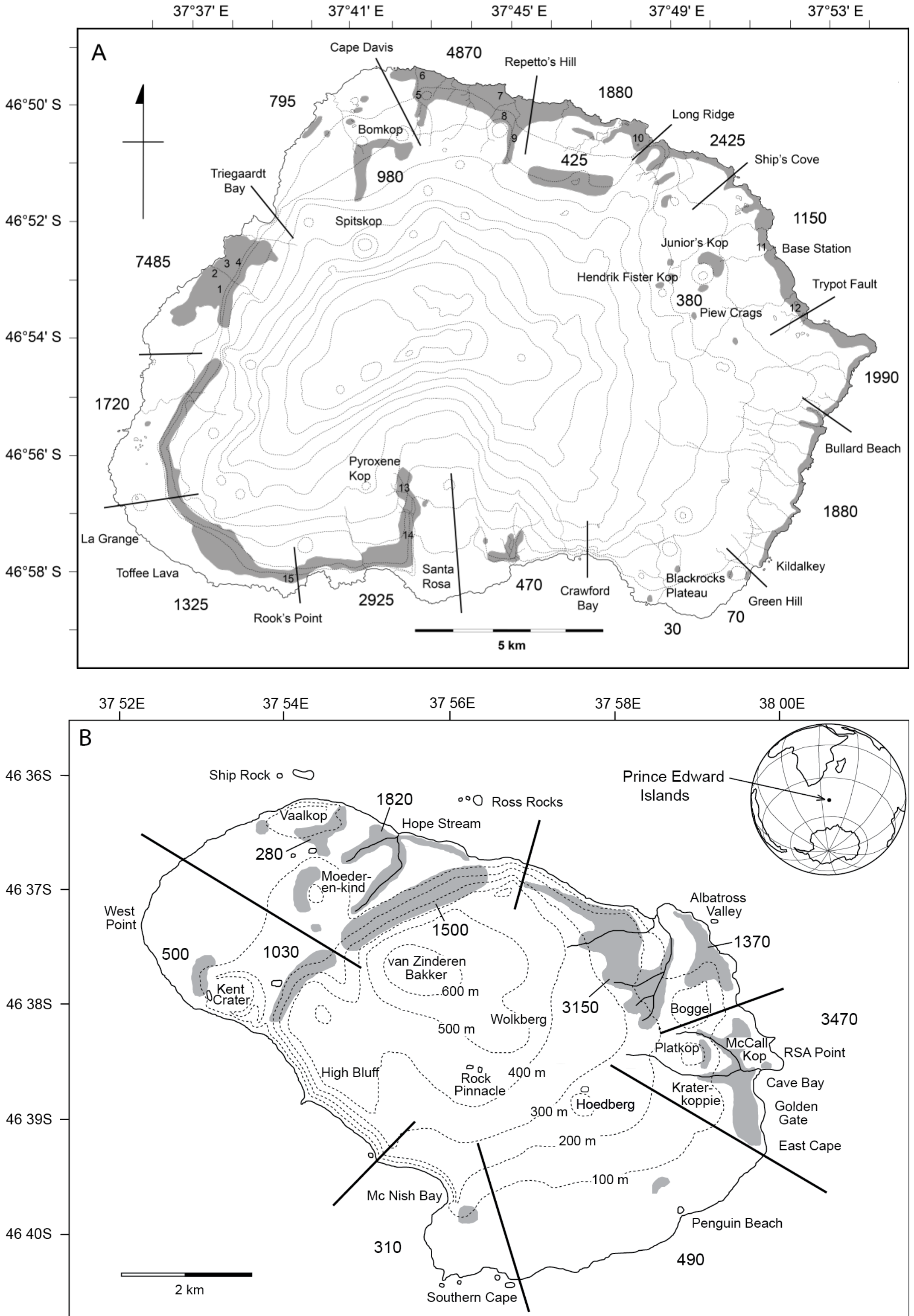

Figure A2.1. The approximate distribution of white-chinned petrel burrows (shaded) and crude estimates of the numbers of burrows (not corrected for observer bias) at Marion Island (**a**) and Prince Edward Island (**b**). Numbered sites on Marion Island indicate where burrow occupancy checks were conducted. The inset in map b shows the location of the Prince Edward Islands.

Also, roughly 15% of pairs do not breed each year (Martin *et al.* 2009), so the total adult population probably is closer to 30,000 pairs.

White-chinned petrels were more patchily distributed at Prince Edward Island than at Marion Island (Fig. A2.1b), due to the predominance of relatively recent black lava flows in most of the coastal lowlands and associated shallow, rocky soils (Verwoerd 1971). Burrows mainly occurred in the northeast of the island, along the top of the northern scarp east to Albatross Valley and along adjacent river valleys. We estimated a total of roughly 14,700 burrows (Fig. A2.1b), which suggests that Prince Edward Island supports some 10,500–12,500 pairs of white-chinned petrels, assuming the same correction factors as Marion Island (95 % CI 9,600–13,500). However, given greater uncertainty about coverage and correction factors, we suggest a broader confidence interval of 9,000–15,000 pairs for Prince Edward Island. Together with Marion Island, this gives an annual breeding population

Table A2.1. The proportion of white-chinned petrel burrows with large entrance moats and one-off burrow occupancy estimates during the early incubation period at 15 sites around Marion Island in 2009. Percentage active is the proportion of burrows that showed signs of recent activity; % occupied is minimum occupancy based on playback response and burrow probing.

Location (numbered 1–15 in Fig. A2.1)	***n* burrows**	**% moats**	**% active**	**% occupied**
1 km south of Mixed Pickle hut (1)	36	53	94	75
Mixed Pickle coastal (2)	59	90	100	81
Mixed Pickle hut (3)	48	15	100	88
Mixed Pickle slope at 82 m (4)	46	39	89	67
Above Cape Davis hut (5)	44	0	66	43
Cape Davis hut to beach (6)	31	0	58	52
Storm Petrel Bay coastal slopes (7)	50	28	86	78
Repettos Hill coastal slope at 112 m (8)	50	26	86	54
Repettos Hill above hut at 185 m (9)	39	5	79	56
Sea Elephant Bay coast (10)	46	41	83	57
Base to van den Boogaard River (11)	50	42	88	66
Trypot Fault (12)	50	40	86	64
Below Pyroxene Kop at 260 m (13)	45	11	91	56
Grey-headed Albatross Ridge (14)	35	26	89	80
Below Rooks hut (15)	36	81	97	47
Total	**665**	**34.4**	**86.9**	**65.0**

for the Prince Edward Islands of roughly 35,000 (29,000–43,000) occupied burrows (Table A2.2) and a total adult population of 40,000 (33,000–50,000) pairs. Together with the 2008 Prince Edward Island estimate (10,500–12,500 pairs, 95% CI 9,600–13,500; details in Ryan *et al*. 2012), this gives an annual breeding population for the Prince Edward Islands of roughly 35,000 (29,000–43,000) occupied burrows (Table A2.2) and a total adult population of 40,000 (33,000–50,000) pairs.

Global population estimate

Combining these results with other recent surveys, the nominate form of the white-chinned petrel occupies close to one million burrows each year (95% CI roughly 0.65–1.3 million burrows; Table A2.2). This estimate excludes pairs whose breeding attempts fail and leave the colony prior to burrow occupancy checks. It also fails to take into account the proportion of pairs that do not breed each year, which may be approximately 15% of pairs (Martin *et al.* 2009). Correcting for the latter factor gives a total adult population of around 1.1 million pairs (0.7–1.5 million pairs). The Prince Edward Islands support the third largest population, after South Georgia and Kerguelen.

Table A2.2. Best estimates of breeding populations of nominate white-chinned petrels, with information on the method used and year of estimate (not applicable for extrapolations based solely on inferred habitat). Estimates rounded to the nearest 100 pairs at most sites.

Island	Occupied burrows (95% CI)	Method	Year	Source
Falklands/Malvinas	70 (55–83)	BC	2006	Reid *et al.* 2006
South Georgia	681,000 (453,000–909,000)[a]	LT	2006/7	Martin *et al.* 2009
Prince Edward Islands	36,000 (29,000–43,000)			Ryan *et al.* 2012
Marion Island	36,000 (29,000–43,000)	BE	2009	Ryan *et al.* 2012
Prince Edward Island	11,500 (9,000–15,000)	BE	2008–11	Ryan *et al.* 2012
Crozet archipelago	23,600 (9,800–36,800)			Barbraud *et al.* 2008
Ile de la Possession	5,800 (5,500–6,100)	BC	2004	Barbraud *et al.* 2008
Other islands	17,800 (4,300–30,700)	Ext H	–	Barbraud *et al.* 2008
Kerguelen archipelago	234,000 (186,000–297,000)			Barbraud *et al.* 2009
Eastern sector	74,000 (58,000–95,000)	LT	2005	Barbraud *et al.* 2009
Remainder of archipelago	160,000 (128,000–202,000)	Ext H	–	Barbraud *et al.* 2009
Total	**974,200 (678,000–1,286,000)**			

Estimates are rounded to the nearest 100 occupied burrows at most sites

BC burrow count, *BE* burrow estimate, *LT* line transect, *Ext H* extrapolation from suitable habitat. Numbers in parentheses give the total number of burrows sampled by random transects (occupied burrows only at South Georgia)

[a] Martin *et al.* (2009) increased these values by 13.5% to account for breeding failures prior to the survey, but this assumes that failed birds leave the colony, which is often not the case. Most studies at other sites have not attempted to correct for this factor, so it is omitted here for comparability across studies

Discussion

Historical records

White-chinned petrels were first reported breeding at the Prince Edward Islands by Crawford (Crawford and Serventy 1952) who noted that they were found ‘‘here and there on the lower grassy slopes… mostly singly’’ (p. 76). Rand (1954) stated that they were largely coastal and were seldom found more than 50 m inland. However, Schramm (1986) reported them as far inland as Junior’s Kop, 2 km inland. van Zinderen Bakker (1971) reported that white-chinned petrels were ‘‘found in a nearly continuous belt around the islands and only very rocky and uneven ground was avoided’’ (p. 166–167). The first population estimates were made by Williams *et al.* (1979), who estimated ‘‘tens of thousands’’ of pairs on Marion Island and Cooper and Brown (1990), who estimated there were ‘‘thousands’’ of pairs on Prince Edward Island. Ryan and Bester (2008) refined this to 20,000 pairs on Marion and 10,000 pairs on Prince Edward Island. These estimates based on field experience proved to be reasonably accurate, only underestimating the best estimates from this study by 15–20%.

Survey techniques used at large sub-Antarctic Islands

Random

Various approaches have been used to estimate white-chinned petrel populations. At very large islands, two random sampling strategies have been used to estimate burrow densities:

Martin *et al.* (2009) counted the numbers of burrows in *6-m diameter circular plots* (28.27 m^2) spaced 10 m apart along straight line transects through tussock-dominated habitat in eight stratified zones at South Georgia. Because their sampling was random and they only reported numbers of occupied burrows, their density data cannot be compared directly with my data. Most plots (94%) in South Georgia contained no occupied nests (Martin *et al.* 2009), whereas my sampling at Marion Island was confined to areas where white-chinned petrels were known to occur. The density at South Georgia in sample plots that contained at least one active nest is roughly 0.10 nests·m^{-2} (Martin *et al.* 2009), which is similar to the density at Marion Island (0.12 active nests·m^{-2}, given an occupancy of rate of 73%). The random circular plot approach used at South Georgia assumed that white-chinned petrel burrows are confined to tussock-dominated vegetation (Martin *et al.* 2009). Although many white-chinned petrel burrows are associated with the tussock grass *Poa cookii* at Marion Island, burrows occur in at least five of the eight main vegetation types (Schramm 1986).

Barbraud *et al.* (2009) avoided the problem of habitat-specific sampling by using *distance sampling along line transects* to estimate burrow densities in 18 stratified zones in the eastern sector of Kerguelen. Such an approach may be the best option for areas that are too large to visit all sites, but for smaller islands a complete census is better, given the patchy distribution of colonies and resultant large variances obtained by random sampling approaches (Schramm 1986; Barbraud *et al.* 2008, 2009; Martin *et al.* 2009).

Systematic

Barbraud *et al.* (2008) counted individual burrows on Ile de la Possession (150 km^2) in the Crozets to demonstrate a decrease in population size between 1983 and 2004. They used a double observer approach at 16 control sites in 2004 to show that burrow detection probability was high (0.92 ± 0.02; Barbraud *et al.* 2008). However, a complete count of individual burrows is not sensible for large populations as it is unrealistic for observers to check thousands of putative burrows to check whether they have collapsed, whether there are multiple entrances to the same burrow, or whether a single entrance serves more than one nest.

The rapid survey technique used in this study on Marion Island is a practical solution to these issues. The number of burrow entrances is estimated at all sites and then corrected by comparing burrow estimates with nest numbers at control sites. Correction factors can be obtained for different observers (to account for individual differences) and different habitats (if burrow detection rate differs between habitats), and the variance in estimates provides a measure of count accuracy. Both techniques fail to account for cryptic nests (e.g., those in rock crevices) or those in areas not searched, and thus estimates are conservative (cf. Ryan *et al.* 2006). Estimating population sizes at islands that are seldom visited poses additional problems. At the Crozet Islands, Barbraud *et al.* (2008) used a simple habitat model based on their observations at Ile de la Possession to predict the area occupied by white-chinned petrels at the other main islands in the archipelago. They assumed that petrels were confined to vegetated areas below 200 m elevation with 5–30° slopes. Had I used such a model for Marion, I would have obtained misleading results, as shown in Chapter 2.

Monitoring populations

The best way to assess the health of white-chinned petrel populations is to monitor well-defined populations regularly (e.g., every five years). Such monitoring should be conducted at sites with reasonably large populations that are readily accessible and that have the most robust baseline estimates, such as at least parts of the populations of Bird Island (South Georgia), Ile de la Possession, and Marion Island. Given the importance of South Georgia for white-chinned petrels, monitoring on part of the main island would be valuable to assess the recovery of this population now that the islands have been declared rodent-free (May 2018, South Georgia Heritage Trust, www.sght.org). A repeatable survey using either burrow counts of selected colonies or fixed transects (to reduce variance among repeat samples) should be set up at one or more sites on the main island. Increasing confidence around estimates for crudely extrapolated populations (e.g., western Kerguelen and islands other than Ile de la Possession in the Crozets) also would be valuable.

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Appendix 3

Trends and tactics of mouse predation on Tristan albatross *Diomedea dabbenena* chicks at Gough Island, South Atlantic Ocean

A pair of Tristan albatrosses *Diomedea dabbenena* return to their six week old chick to find it has died from its mouse inflicted wounds, Gough Island 2014 (photo Delia Davies, remote camera).

This appendix is broadly based on this publication:

Davies, D., Dilley, B., Bond, A., Cuthbert, R. and Ryan, P., 2015. Trends and tactics of mouse predation on Tristan Albatross *Diomedea dabbenena* chicks at Gough Island, South Atlantic Ocean. *Avian Conservation and Ecology*, 10(1).

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Appendix 3: Trends and tactics of mouse predation on Tristan albatross *Diomedea dabbenena* chicks at Gough Island, South Atlantic Ocean

Abstract

The critically endangered Tristan albatross *Diomedea dabbenena* breeds almost exclusively on Gough Island, in the central South Atlantic, where breeding success is much lower than other great albatrosses *Diomedea* spp. worldwide. Most breeding failures occur during the chick-rearing stage, when other great albatrosses suffer few failures. This unusual pattern of breeding failure is assumed to be largely due to predation by introduced house mice *Mus musculus*, but there have been few direct observations of mouse attacks. We closely monitored the fates of 20 chicks in the Gonydale study colony (123 chicks in 2014) using motion-activated cameras to determine the causes of chick mortality. Only 5 of 20 chicks survived to fledge, and of the 15 failures, 14 (93%) were due to mouse predation. One mouse-wounded chick was killed by a southern giant petrel *Macronectes giganteus*; the rest died outright from their wounds within 3.9 ± 1.2 days of the first attack. Despite this high impact, most chicks were attacked by only 1–2 mice at once (maximum 9). The remaining 103 chicks in the study colony were checked less frequently, but the timing of failures was broadly similar to the 20 closely monitored nests, and the presence of mouse wounds on other chicks strongly suggests that mice were responsible for most chick deaths. Breeding success in the Gonydale study colony averages 28% from 2001 to 2014; far lower than the normal range of breeding success of *Diomedea* species occurring on islands free from introduced predators. Island-wide breeding success fell below 10% for the first time in 2014, making it even more urgent to eradicate mice from Gough Island.

Introduction

Many seabirds nest on isolated islands that lack land mammals (Schlaepfer 2002) and consequently they are particularly susceptible to the introduction of mammalian predators such as cats *Felis catus* and rodents (Atkinson 1985; Croxall *et al*. 2012). Understanding the effects of introduced mammals on island seabirds are a key issue for conservation biologists (Jones *et al*. 2008; Le Corre 2008; Jones and Ryan 2010; Medina *et al*. 2011), and Gough Island, in the central South Atlantic, is a poignant example of an introduced mammal, the house mouse *Mus musculus*, dramatically affecting the breeding success of a suite of seabirds by preying on their chicks (Cuthbert *et al*. 2013, 2014).

Gough Island (40˚82' S, 9˚85' W) is a UNESCO World Heritage Site and an Important Bird Area (IBA) that hosts significant populations of several globally threatened species. Mice are the only introduced mammal on Gough Island. They were brought to the island inadvertently by sealers in the 19th century, and are now ubiquitous throughout the island (Rowe-Rowe and Crafford 1992). Initially, mice were considered to have little impact on the island's birds, being regarded as 'probably harmless' (Elliott 1953). However, observations in 2001 led Cuthbert and Hilton (2004) to propose mouse predation as the most probable cause of the unexpectedly high breeding failure of Tristan albatrosses *Diomedea dabbenena* and Atlantic petrels *Pterodroma incerta* on Gough Island. Subsequent studies confirmed that mice kill the chicks of a wide range of seabirds and also threaten Gough's endemic bunting *Rowettia goughensis* (Wanless *et al*. 2007; Ryan and Cuthbert 2008; Cuthbert *et al*. 2013).

The population of Tristan albatrosses breeds almost entirely on Gough Island (~1800 pairs, Cuthbert *et al*. 2014), with only 1–2 pairs on Inaccessible Island in the nearby Tristan da Cunha archipelago (Ryan 2005). The population is decreasing by approximately 3% per year (Wanless *et al*. 2009; Cuthbert *et al*. 2014), resulting in the species being listed as Critically Endangered (IUCN 2014). The decline is being driven by the combination of mortality of birds caught on fishing gear at sea and low reproductive output from mouse predation (Wanless *et al*. 2009). Breeding success is much lower than that of *Diomedea* spp. albatrosses breeding at predator-free locations (Croxall *et al*. 1990; Weimerskirch 1992; Tickell 2000; Nel *et al*. 2003). The majority of breeding failures occur during the chick-rearing period and are assumed to be largely due to predation by introduced house mice (Wanless *et al*. 2007).

Most evidence of mouse impacts on Tristan albatrosses comes from chicks showing wounds characteristic of mouse attacks (Wanless *et al*. 2007). There have been few direct observations of mouse attacks on albatross chicks, because mice are active at night. It is not known how quickly mice kill albatross chicks, or indeed whether they kill the chicks directly or merely weaken them to the point where they fall prey to other predators such as southern giant petrels *Macronectes giganteus* or brown skuas *Stercorarius antarcticus* (Wanless *et al*. 2009). The aims of this study were to assess mouse predation on Tristan albatross chicks by monitoring and filming a sub-sample of 20 chicks intensively. We were particularly interested in the proportion of chicks that failed as a result of mouse predation, the mechanisms of mouse predation, and the final outcome of mouse attacks.

Methods

Tristan albatrosses lay eggs in late December-January, with chicks fledging in November. Breeding success of the entire population has been estimated since 2001 by conducting annual counts of incubating adults in late January/February and of large chicks in September (excluding 2002, 2003, 2005 and 2011; Cuthbert *et al*. 2014; Fitzpatrick Inst. unpublished data). Counts were divided into 8–12 geographical zones across the island. Since 2008 more accurate estimates of breeding success have been obtained from a study colony in the Gonydale Valley (c. 150–190 nests per year) where nests and birds were individually marked by researchers who are based on the island year round. Nests were checked approximately every four weeks from laying to fledging (Cuthbert *et al*. 2014), although in 2014 checks were more frequent (about every 5 days around laying and hatching, and approximately every 2 weeks at other periods). Date of laying was taken as the mid-point between nest checks and date of hatching was taken from when chicks were almost fully out of their eggs or were still wet after hatching (15 nest checks over 79 days). Newly pipped eggs were excluded as the time to emerge varied considerably; two chicks took more than 6 days to fully emerge from their eggs once pipping began.

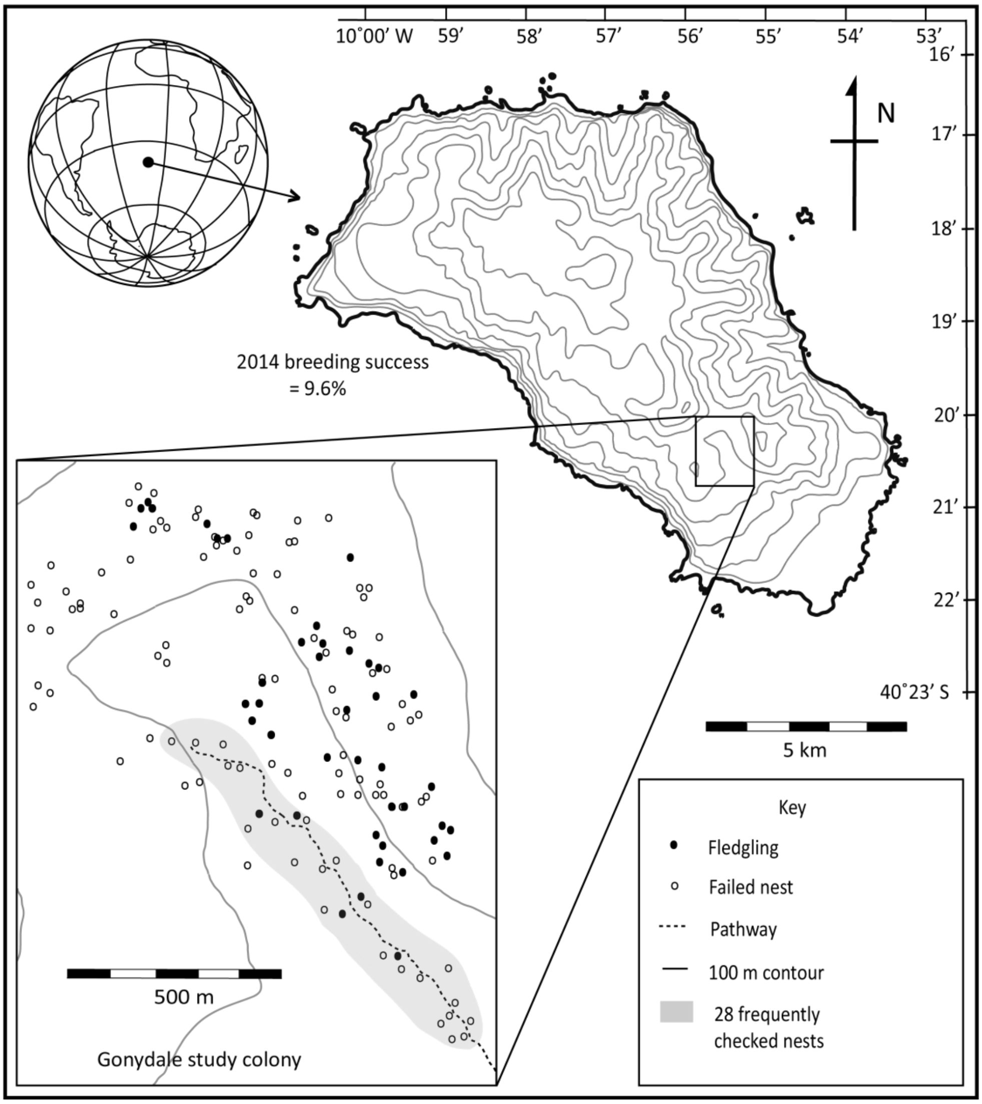

Figure A3.1. Gough Island showing the location of the long-term Tristan albatross *Diomedea dabbenena* study colony in the Gonydale Valley. Inset shows the location of all nests in 2014 and the shaded area along the pathway where 28 nests were monitored frequently during the early chick period.

In order to gain a better idea of the causes of breeding failure, 28 nests along a path within the Gonydale study colony (Fig. A3.1) were checked until hatching, with the 20 hatched chicks checked frequently from after hatching (29 March) to mid-winter (8 July 2014; 22 checks, 4.7 ± 4.0 days between checks). To estimate when chicks hatched, were left alone or died, we assumed these events occurred at the mid-point between successive checks. On each visit, chicks were inspected for wounds, typical of those inflicted by mice (Wanless *et al*. 2007, 2009, 2012, Jones and Ryan 2010). The cause of these wounds was confirmed by filming the nests with motion-activated trap cameras (Bushnell Trophy Camera, model 119436) that recorded nocturnal activity with infra-red images. The infra-red flash does not deter predators and these cameras have been used to record northern giant petrels *Macronectes halli* predating on wandering albatross *Diomedea exulans* chicks on Marion Island (Dilley *et al*. 2013). Cameras were mounted 30 cm above the ground on PVC poles, 4–5 m from the nest, and set on high motion sensitivity to take one image per second for 3 seconds upon activation. We used external 6V

12 amp-hour batteries (insulated against the cold weather in plastic tubs) which allowed cameras to run for up to 14 days. By rotating three trap cameras we were able to monitor eight chicks. One chick had a small mouse wound when filming began, but the remaining seven chicks were not wounded. However, chicks were selected based on proximity to other wounded or recently killed chicks because mouse attacks tend to be spatially coherent (Wanless 2007). Detailed behaviour of mice attacking wounded chicks at night was recorded by direct observations, and with a GoPro Hero3 video camera using an external red light source.

Analyses of breeding success trends were conducted in the R statistical environment (R Core Team 2014) where we used a binomial generalised linear model run in package lme4 (Bates *et al*. 2014). The statistical power to infer a decrease in breeding success since 2001 was estimated based on the linear regression of reproductive success over time. We also estimated the overall linear population trend using TRIM 3.54 (Pannekoek and van Strien 2001) using counts of incubating adults accounting for serial correlation. As we counted the entire population, we did not adjust for overdispersion. The multiplicative rate of increase (λ) is presented ± SE (Cuthbert *et al*. 2014). All other means are presented ± SD.

Results

Laying, hatching and incubation period

In Gonydale, peak egg laying occurred in the last week of December and early January, with 66% (82/123) of eggs laid by 6 January 2014, 95% by 15 January and the last eggs laid by 31 January 2014. For the 28 closely monitored nests, hatching occurred from 13–28 March 2014 (22 March 2014 ± 4.6 days, *n* = 20). Hatching was more protracted across the whole study colony: 8 March to 6 April 2014 (23 March 2014 ± 6.3 days, *n* = 123). Incubation lasted 73–80 days (75.9 ± 1.8 days, *n* = 13), similar to other great albatrosses (average 78–79 days, Tickell 2000). For these 13 nests, laying date was accurate to within 3 days and at hatching, the chicks were observed in the advanced stages of hatching (n = 6) or were still wet (n = 7).

Causes of breeding failure

Of the 28 frequently monitored nests along the Gonydale path, 8 failed at the egg stage (29%), slightly more than the entire Gonydale colony in 2014 (17%, *n* = 149). Of the 20 chicks that hatched, one small chick disappeared overnight while being brooded, when its nest mound was partly washed away in heavy rains. The other 19 chicks survived the brood-guard phase and were left by their parents 33 ± 6.8 days after hatching (range 20–49 days, *n* = 19). Although Wanless *et al*. (2007) reported wounded chicks in March, when still being brooded, the first wounded chick in 2014 was observed on 1 May, 9 days after the parents had left it on its own. Chicks were first attacked by mice 30 ± 20.4 days after being left alone (range 1–70 days, *n* = 16).

Overall, 16 of 19 (84%) monitored chicks were attacked by mice, with attacks starting when the chicks were 64 ± 19 days old (range 35–102 days, *n* = 16). Of these 16 wounded chicks, 2 survived, but 14 (88%) died within 3.9 ± 1.2 days (range 2–5 days) of first being attacked. Of these 14 chicks, six were definitely killed by mice (evidence from cameras; Table A3.1) and eight died shortly after first showing signs of mouse attacks and were almost certainly killed by mice.

Table A3.1. Summary of camera effort (61.8 days) monitoring eight Tristan albatross *Diomedea dabbenena* chicks where six chicks die from mice wounds, one is attacked but recovers (nest 5), and one is not attacked (nest 8). For nest 2, the first attack by mice was not filmed (numbers in parentheses excluded from mean ± SD).

Camera nest	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**Mean ± SD**
Time monitored (days)	3.5	1.1	3.3	12.5	14.0	14.6	2.8	10.0	7.7 ± 5.6
Nights when mice attacked	4	(3)	3	9	2	4	4	0	3.7 ± 2.7
Number of mice in first attack	1	-	1	1	1	1	1	0	1.0 ± 0.0
Maximum number of mice per night	9	2	2	2	1	2	2	0	2.0 ± 1.3
Average maximum mice per attack night	5.0	1.7	1.3	1.2	1.0	1.8	2.0	0	2.0 ± 1.3
Days from first attack to death	3.3	(2.7)	4.1	2.6	-	3.2	5.0	-	3.5 ± 0.9

Cameras recorded activity at eight Tristan albatross nests for a total of 61.8 days from 1 May to 24 July 2014 (Table A3.1). Cameras captured a total of 98,283 photos (10,920 ± 10,115 photos per nest), with cameras triggered more frequently at nests where chicks were more active due to mouse disturbance. In addition, we recorded a total of 15.0 hours of GoPro video footage from 4 nests and 8.0 hours (over two nights) of direct observations at one nest (Fig. A3.2).

Figure A3.2. A six week old Tristan albatross *Diomedea dabbenena* chick is attacked by mice despite the parent being present at the nest. This chick died 3.3 days after the first mouse attack (Photo Ben Dilley).

Only one of the filmed chicks was not attacked by mice (camera nest 8, Table A3.1). Mice attacked chicks on 29 of the 62 film nights (47%). All attacks happened at night and the initial attack was always by a single mouse, which repeatedly targeted the same site, usually on the chick's rump, clinging to its down and biting until the skin was penetrated. When displaced by the irritated chick nuzzling its wound, or vigorously shaking its body, the mouse would return to the wound within seconds. Chicks were seldom attacked by more than 1–2 mice at a time, although one chick had up to 9 mice attacking at once (Table A3.1). The behaviour of the mice suggested that this was a well practiced technique, as they generally climbed directly onto the nest mound and the chick without hesitation.

Chicks would often nuzzle at their open wound and so accumulate blood on their bill which the mice would then proceed to lick off, often with little or no protest from the injured chick (Fig. A3.2). This fearless behaviour by the mice did not alter if an albatross parent was present at the nest (Fig. A3.2), or in one case, even brooding its chick. The parents made little, if any, attempt to scare away the mice. In one case a male parent seemingly attempted to protect its five-week old chick, which had a large open wound on the back of its neck. The mice continued to feed on the chick despite the parent's attempt to brood. It was noticeable that mouse attacks greatly increased activity by albatross chicks at night. Without mice present, albatross chicks slept with their heads tucked in, seldom moving at all. By comparison, chicks attacked by mice were unable to sleep, repeatedly trying to chase off the mice, although these efforts became weaker on successive nights.

Of the seven chicks attacked by mice (Table A3.1), five died outright from their wounds (Fig. A3.3); one severely wounded chick was killed by a southern giant petrel and one chick recovered from its wounds. One of the chick fatalities (camera nest 6, Table A3.1) had a relatively small wound compared to other fatally wounded chicks (camera nests, *n* = 6 wounded) and had not been fed for 10 days when it eventually died after a period of cold weather. The chick that survived its wounds (camera nest 5, Table A3.1) was fed by its parent in the 24 hours before its first attack.

Figure A3.3. Five of the seven filmed chicks that were attacked by mice died outright from their wounds within 3.9 ± 1.2 days of the first attack. Here the parents return to feed their chick to find it has died since their last visit (Photo Delia Davies, remote camera).

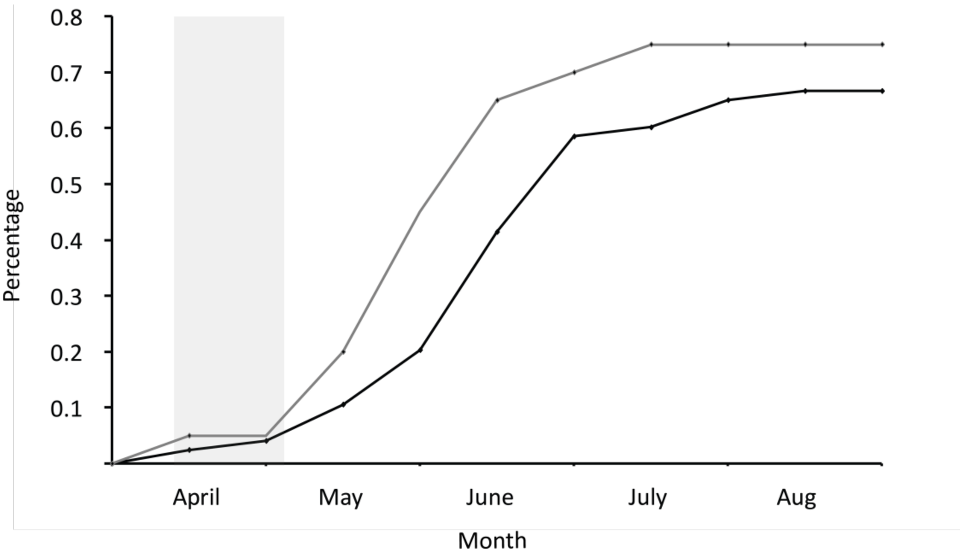

Figure A3.4. The timing of Tristan albatross *Diomedea dabbenena* chick failures in the Gonydale study colony (103 chicks, black) and for the subsample of frequently checked chicks within the colony (20 chicks, grey) in 2014. Chick mortality represented as a cumulative percentage (y axis). The shaded area represents the time when chicks are first left alone.

For the entire Gonydale study colony, overall breeding success was 28% (n = 149 incubating pairs). Most chick failures (78%) occurred in the first three months after hatching (64/82 chick failures, 0.69 chicks per day). Although these nests were checked less frequently, the timing of failures synchronised broadly with the 20 frequently checked nests (Fig. A3.4) and is consistent with the

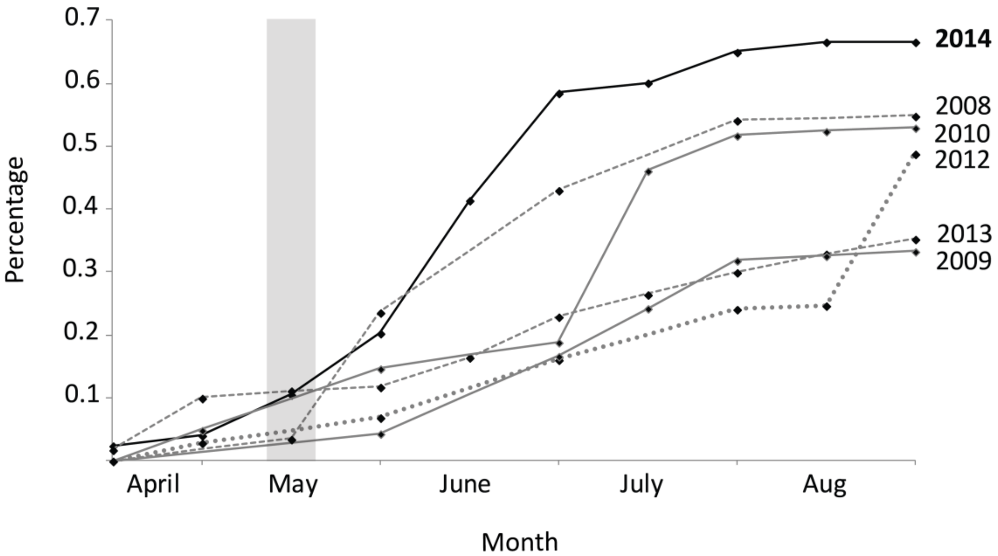

Figure A3.5. Tristan albatross *Diomedea dabbenena* chick failure rate for the years 2008 (n = 161 nests), 2009 (n = 172), 2010 (n = 175), 2012 (n = 192), 2013 (n = 201), and 2014 (n = 149) in the Gonydale study colony. No data for 2011. The shaded area represents the time when chicks are first left alone.

timing of failures in this colony from 2008 to 2013 (Fig. A3.5). The most common site for chicks to be wounded was the lower rump (n = 6), but mice also targeted the back of the neck (2), top of the head (1), side of the lower mandible (1), the wings (1), or a combination of these locations (1). To view a one minute video of mice attacking a Tristan albatross chick follow this link: [online] URL:http://youtu.be/XNxqIYLthus.

Trends in breeding success and overall population

The 2014 breeding success for the island population was 9.6% (n = 1704 incubating pairs, Table A3.2), the lowest recorded since annual island-wide counts started in 2001 (Fig. A3.6). Island-wide breeding success from 2001 to 2014 averaged 28.3 ± 12.7%.

Table A3.2. Island wide Tristan albatross *Diomedea dabbenena* counts for years 2010 – 2014.

Year	2010	2011	2012	2013	2014
Incubators	1698	-	1421	1748	1704
Large chicks	261	360	482	578	163
Island breeding success	15.4%	-	33.9%	33.1%	9.6%

The regression slope (β = -0.025 ± 0.406) was not significantly different from 0 (t = -1.1, p = 0.26, *n* = 10 years), but the power to detect a significant decrease of this magnitude is small (~0.26) given the relatively short sampling period (Fig. A3.6). The highest recorded chick production was in 1999, when almost twice as many chicks were counted as in any other year (Wanless *et al*. 2009). The 163 chicks counted across Gough Island in September 2014 were only 14% of the 1129 chicks counted in September 1999 (Ryan *et al*. 2001). The overall population of incubating Tristan Albatrosses is decreasing (λ = 0.979 ± 0.002, Wald χ^2_1 = 137.07, p < 0.01), and currently numbers around 1650 breeding pairs breeding annually (range from last three censuses: 1451–1745), a decrease of approximately 150 pairs since 2011 (Cuthbert *et al*. 2014).

Discussion

Despite albatross chicks being two orders of magnitude larger than house mice (Cuthbert and Hilton 2004, Wanless *et al*. 2007), mice were responsible for 93% (14/15) of Tristan albatross chick failures in an area with higher than average breeding success monitored on Gough Island in 2014. Although mice were suspected of attacking Tristan albatross chicks on Gough Island in 2001 (Cuthbert and Hilton 2004) and this was confirmed in 2004 (Wanless *et al*. 2007), ours are the first definite records of chicks being killed outright by mice. It was reasonably assumed that most chicks injured by mice were killed by southern giant petrels and brown skuas because these birds are frequently observed feeding on freshly dead albatross chick carcasses (Verrill 1895; Wanless *et al*. 2009). However, of the six chick fatalities filmed, five died outright from their wounds and only one wounded chick was killed by a southern giant petrel. This suggests that in most cases these birds scavenge chicks killed by mice. The actual cause of death appears to be a combination of physical injury and exhaustion from the absence of sleep and physical effort of being disturbed repeatedly by mice at night. Wounds observed on chicks elsewhere in the study colony closely resembled those seen in chicks filmed being attacked by mice. This fact, together with the presence of mouse wounds on chicks, strongly suggests that mice are largely responsible for the low breeding success of Tristan albatrosses.

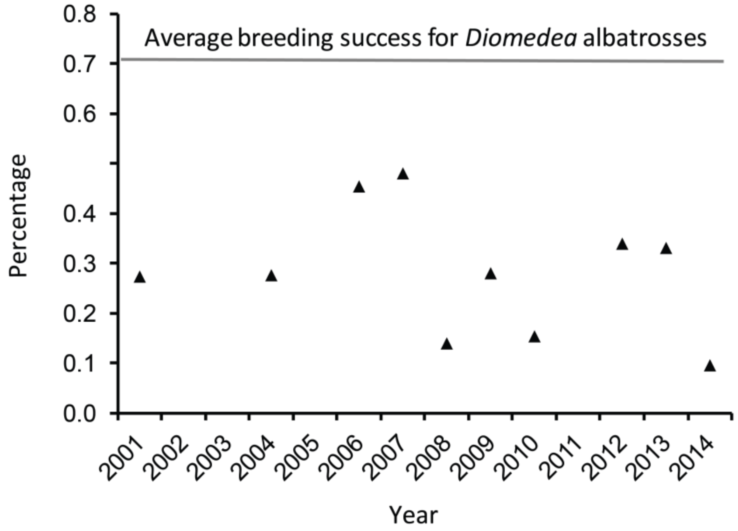

Figure A3.6. Estimates of island wide breeding success for the Tristan albatross *Diomedea dabbenena* from 2001 – 2014 (n = 10 years; total number of island wide incubating pairs ranged from 1271 – 2400; no data for years 2002, 2003, 2005, 2011) compared to the average breeding success for *Diomedea* spp. albatrosses (Tickell 2000).

Our other significant finding was how quickly the process happens from the initial mouse attack to the death of the chick (3.9 ± 1.2 days). This explains why relatively few injured chicks were recorded during monthly nest checks compared to the number of eventual failures. Most chicks simply disappeared between colony checks, with few clues as to the cause of death. Predation rates on albatross chicks peaked in May-June (Figs. A3.4 and A3.5), at the start of the austral winter, when mice turned to birds as an alternative food source. Cuthbert *et al*. (2016) found that mice sampled within Tristan albatross breeding areas showed an increase in mass and body condition over this same period.

Because mice are the only terrestrial mammal introduced to Gough Island, they occur at very high densities of up to 266 mice·ha^{-1} (Cuthbert *et al*. 2016). The lack of mammalian predators and competitors, together with the abundance of albatross and petrel chicks in winter, seemingly has encouraged the mice to adopt predatory behaviour (Cuthbert *et al*. 2016). It was noticeable that in 2014 virtually the only chicks to survive in areas of very high albatross mortality (e.g., West Point) were found at the highest elevations, suggesting that mouse attacks were worse in more moderate climates where mouse numbers may remain higher in winter. It has already been argued that global warming may further benefit mouse populations to the detriment of breeding seabirds (Huyser *et al*. 2000; Millien 2006; de Villiers and Cooper 2008; Ferreira *et al*. 2006).

Based on complete island counts of all incubating pairs and fledglings, the island wide breeding success of the Tristan albatross since 2001 has been very low and averaged around 28%; far lower than average breeding success for *Diomedea* spp. albatrosses elsewhere (Tickell 2000) (Fig. A3.6). In addition, 2014 had the lowest breeding success yet recorded for both the whole island counts and the Gonydale study colony (Cuthbert *et al*. 2014). Although rodents and albatrosses co-exist on other oceanic islands, the level of mouse predation we observed appears to occur only on Gough. Incidents of mouse-injured albatross have been reported from Marion Island (Jones and Ryan 2010) where peak mouse densities (237 mice·ha^{-1} in mire habitats, McClelland 2013) are similar to those on Gough (266 mice·ha^{-1}, Cuthbert *et al*. 2016). However, the wandering albatross *Diomedea exulans* population on Marion is stable (Nel *et al*. 2003; ACAP 2015) and incidents of mouse injured albatross chicks are rare (Jones and Ryan 2010; Dilley *et al*. 2013; but see Chapter 6). By contrast, incidental sightings of mouse-injured Tristan albatross chicks are common on Gough and mouse predation has an island-wide impact on chick production.

On Amsterdam Island, the breeding success of the critically endangered Amsterdam albatross *Diomedea amsterdamensis* is low (61%, Rivalan *et al*. 2010) relative to wandering albatross in the Indian Ocean (Weimerskirch *et al*. 1997). Thiebot *et al*. (2014) investigated the effect of introduced mammals (cats, rats *Rattus* spp. and mice) on brooded chicks by monitoring 12 nests with camera traps (25,000 photos over 2 months), but did not record any predation or interaction event and could not relate the rate of breeding success to chick predation.

In summary, the Tristan albatross population cannot sustain current levels of chick mortality (Wanless *et al*. 2007, 2009); any further increase in mouse attacks will only accelerate the rate of population decrease. Fortunately it is possible to eradicate invasive rodents from islands (Clout and Veitch 2002; Angel *et al*. 2009). In the past, failure rates of mouse eradication attempts have generally been higher than for eradicating rats (Howald 2007; Cuthbert *et al*. 2011), but the recent successful eradication of rabbits, rats and mice from Macquarie Island (~128 km^2, Tasmania Parks and Wildlife Service, 2014) is extremely encouraging given that Gough Island is about half the size (~65 km^2). A mouse eradication programme on Gough Island for the conservation of the Tristan Albatross and other vulnerable birds is a top conservation priority (Dawson *et al*. 2014).

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