POPULATION TRENDS OF SPECTACLED PETRELS *PROCELLARIA CONSPICILLATA* AND OTHER SEABIRDS AT INACCESSIBLE ISLAND

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ABSTRACT

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Inaccessible Island, in the Tristan da Cunha archipelago, is the sole breeding site of the Spectacled Petrel *Procellaria conspicillata*. The island also supports globally important populations of four threatened seabirds, as well as populations of other seabird species. A seabird monitoring protocol was established in 2004, following baseline surveys of most surface-breeding species in 1999. For the species monitored, we report population trends that are based on visits in 2009 and 2018. Populations of most monitored species appear to be stable or increasing, including three albatross species currently listed as Endangered or Critically Endangered. However, numbers of Northern Rockhopper Penguin *Eudyptes moseleyi* may have decreased slightly since 1999, and numbers of Antarctic Tern *Sterna vittata* have decreased since 1982. The population of Spectacled Petrels is estimated to be at least 30 000 pairs and continues to increase since feral pigs *Sus scrofa* died out on the island in the early 20th century. We describe a new monitoring protocol for Spectacled Petrels that will be easier to repeat and implement and that should provide a more sensitive measure of future population changes.

Key words: Inaccessible Island, Tristan da Cunha, Spectacled Petrels, seabird population trends, feral pigs

INTRODUCTION

Inaccessible Island is a 14 km^2 uninhabited island in the Tristan da Cunha archipelago. It was declared a nature reserve in 1997, a natural World Heritage Site (as part of the Gough and Inaccessible Islands World Heritage Site) in 2004, and as a Ramsar wetland in 2008. The island is the only known breeding site of the Spectacled Petrel *Procellaria conspicillata* (listed as globally Vulnerable; BirdLife International 2018a) and is home to other globally important seabird populations, including three albatross species listed under the Agreement on the Conservation of Albatrosses and Petrels (www.acap.org). The island supports a remnant population of Tristan Albatross *Diomedea dabbenena* (Critically Endangered), the last population remaining in the Tristan archipelago (Ryan *et al.* 2001), and it is the only breeding site for this species that is free from introduced mammalian predators (Cuthbert *et al.* 2004). Inaccessible Island is also one of only four breeding locations for Atlantic Yellow-nosed Albatross *Thalassarche chlororhynchos* (Endangered), and it has a significant population of Sooty Albatross *Phoebetria fusca* (Endangered). Northern Rockhopper Penguins *Eudyptes moseleyi* (Endangered) breed at nine sites on the island (Ryan & Moloney 2000), which supports roughly 22 % of the world population for this species (Robson *et al.* 2011, BirdLife International 2018b).

The status of seabirds at Inaccessible Island was reviewed by Fraser *et al.* (1988) and updated by Ryan *et al.* (1990). Ryan & Moloney (2000) published the most recent overview of the island's seabird populations, based on a three-month visit in 1999/2000. This visit also resulted, in part, in the development of a Seabird Monitoring Manual for the island (Ryan 2005). This manual acknowledged the difficulty of access to the island and provided a set of pragmatic approaches to estimate the status of the island's surface-nesting seabird populations, assuming a visit of several weeks in November (early summer) to Blenden Hall, on the island's west coast. Blenden Hall offers the only ready access to the island plateau, which is where Spectacled Petrels and most albatrosses breed. These monitoring protocols were field tested in November 2004 (Ryan 2005) and resulted in an updated estimate of the Spectacled Petrel population (Ryan *et al.* 2006). Since then, PGR visited the island twice in November during which he monitored and updated the ongoing growth of the Spectacled Petrel population: in 2009 (Ryan & Ronconi 2011) and in 2018 (this paper). Based on these surveys, we report trends in most of the island's surface-nesting seabird populations over the last two decades. We also suggest a revised monitoring approach for Spectacled Petrels that will be easier to implement and that will provide a more sensitive indicator of population change.

METHODS

We visited Inaccessible Island from 13 September to 26 November 2018, working from the field hut at Blenden Hall or from a camp established on the island plateau near Denstone Hill. Details of visits in 2004 and 2009 are presented in Ryan *et al.* (2006) and Ryan & Ronconi (2011), respectively. During each of these visits, we implemented the count methods outlined in the *Inaccessible Island Seabird Monitoring Manual* (see Ryan 2005 for details), although we revised the count approach for Spectacled Petrels in 2018. An additional visit took place from 15 September to 06 October 2011; this was too early to census Spectacled Petrels or summer-breeding albatrosses, but it was well-timed to census incubating Northern Rockhopper Penguins.

Spectacled Petrels

Previous counts of Spectacled Petrels (Rowan *et al.* 1951, Hagen 1952, Fraser *et al.* 1988, Ryan & Moloney 2000, Ryan *et al.* 2006, Ryan & Ronconi 2011) attempted to estimate the entire island population. This was feasible when the population was smaller, given the strongly clustered distribution of burrows, with most pairs breeding in conspicuous terraces created by the petrels in bogfern *Blechnum palmiforme* heath. This latter habitat dominates most of the western plateau of Inaccessible Island (Ryan 2007). However, the continued growth of the population was making a complete count increasingly arduous, and the approach was not sensitive to subtle changes in the species' range or burrow densities because counts were simply reported per catchment area (Ryan *et al.* 2006, Ryan & Ronconi 2011). In 2018, we adopted a more repeatable, transect-based approach that sampled the entire range of the species. We counted all burrows within five metres of either side of a series of north-south transect lines running through the species' known breeding range (Ryan & Ronconi 2011). That is, we conducted a strip survey, where burrow density was the number of burrows found in the sampled area (transect length × 10 m). This method was preferred to a distance-based line transect approach because of the great difference in detection distance between isolated burrows and petrel terraces. We walked transects every 0.1 minutes of longitude (which equates to 147 m between transects) throughout the species' main breeding range, covering 14 lines from 012°40.6′W to 012°41.9′W. We walked transects every 0.05 minutes of longitude (i.e., 73.5 m apart) through the smaller outlying colonies on Denstone Hill (five lines from 012°40.05′W to 012°40.25′W) and Round Hill (six lines from 012°39.75′W to 012°40.0′W; Fig. 1). At these outlying colonies, we worked from a central line outwards, and we considered the first outer transect with zero burrows to be the colony boundary. All transects were counted between 29 October and 15 November 2018.

Most lines were walked by two observers walking roughly five metres apart and guided by a handheld GPS (Garmin 60 and 64s) to remain on the desired track. We recorded the habitat (three main habitats based on the dominant vegetation, see below) along each transect, noting where each habitat type started and ended; this was subjectively determined in the field as a point where the dominant vegetation type changed. We counted the number of burrows in each habitat block, recording the locations of the first and last burrows. For the purposes of mapping the species' range, we combined habitat blocks if they were separated by less than 50 m of unoccupied habitat. Spectacled Petrel burrows are, for the most part, large and fairly easy to detect. However, in some dense tussock slopes, burrows that were difficult to locate were found using playback of a repertoire of Spectacled Petrel groans and rattles (the petrels are very responsive to playback during late October and early November; Ryan *et al.* 2006).

Burrow detection rate and occupancy were checked by assessing the status of 20 burrows at each of five locations during the census period: two in Ringeye Valley, one on Cairn Peak, one at Molly Bog, and one on Round Hill (Fig.1). These checks determined the likelihood that a burrow was indeed a burrow, ensured that no burrows were missed, and determined the occupancy rate of burrows by response to playback. The repertoire described above was played directly into the burrow entrance for 15 seconds using a JBL IPX7 Bluetooth speaker at medium volume. If this failed to elicit a response, the burrow contents were checked by probing

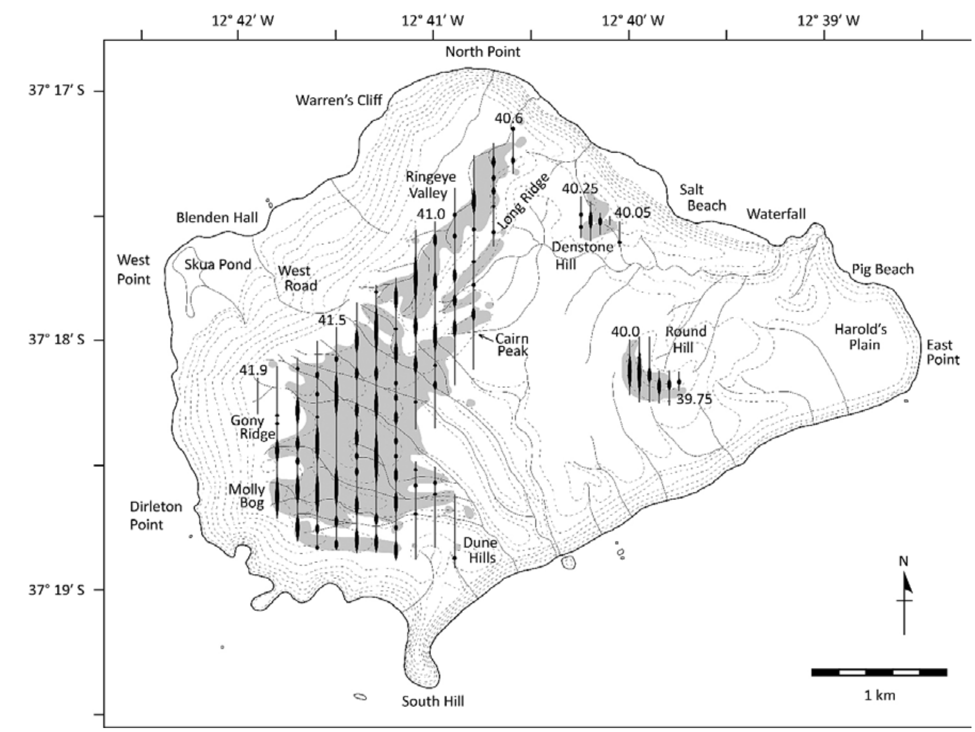

Fig. 1. The distribution of Spectacled Petrel nests counted during transects at Inaccessible Island, superimposed on the range estimated in 2009 (grey shaded area from Ryan & Ronconi 2011). Fine lines indicate transect areas sampled; black bulges indicate occupied patches (i.e., pooling areas with gaps smaller than 50 m between occupied patches).

with a stick or by using a burrowscope. We also counted all burrows within the upper section of Ringeye Valley, as recommended in the monitoring manual (Ryan 2005).

We used two approaches to estimate the total population of Spectacled Petrels: 1) simple extrapolations based on the area sampled, which consisted of 7 % of the main range and 14 % of the peripheral populations on Denstone Hill and Round Hill, and 2) extrapolations based on the average density in the different habitats sampled (i.e., stratified by habitat). The petrels breed in three main types of habitat: *Scirpus* terraces created by the petrels' burrowing activities, bogfern heath, and slopes covered in tussock grass *Spartina arundinacea*. Few birds breed on slopes dominated by ferns other than bogferns or in island tree *Phylica arborea* woodland. Because these habitats support less than 0.5 % of the population (see Results), they were ignored for this exercise. We used Google Earth Pro to estimate the areas of each of the three main habitat types available within the breeding range. However, the Google Earth image was from 2005 and thus didn't reflect the current extent of terraces, which have increased with the petrel population. It was also difficult to identify smaller terraces among bogfern heath in the image. Therefore, we made two extrapolations: one based on three habitat types and one pooling both terraces and bogfern heath (because all terraces were within this habitat type). Populations were extrapolated based on the average ± standard error (SE) of burrow density (burrows per 100 m^2) recorded per habitat type. Burrow counts and extrapolated counts were converted into occupied burrows, which we defined as the best estimate of the number of breeding pairs, based on the burrow detection and occupancy tests described above.

Other seabird species

Northern Rockhopper Penguin populations were crudely estimated in 2009 and 2018 from counts of beach parties at each of the breeding colonies, a procedure that provides a rough proxy of colony size (Ryan *et al.* 1990, Ryan & Moloney 2000, Ryan 2005). Shore-based counts (n = 5–8) were made throughout November for colonies accessible from Blenden Hall, with the remaining colonies counted from images taken from a small boat (03 November) or ship (26 November, 30 November, and 01 December) that passed close to shore around the island's coast. We obtained a correction factor for the vessel-based counts on 01 December, when beach parties at three colonies on the east coast (Waterfall Slump, Salt Beach main, and Salt Beach north; Fig. 1) were photographed from a ship between 12h30 and 13h00, then counted during shoreline walks between 14h30 and 16h00. The beach party at the South Hill Beach also was counted from the cliffs above the beach on 11 November.

Estimates of the number of penguins breeding in each of the two colonies accessible from Blenden Hall were made in 2009 (Robson *et al.* 2011) and were repeated in September 2011. Colony areas were estimated by walking the perimeter with a handheld GPS. Penguin density was then estimated within each colony by counting the number of nests in 25 randomly selected 2×2 m quadrats. Incubation counts were not undertaken in 2018 because many birds abandoned their nests when we tried to map the edge of the colony, and several eggs were lost to predation by Tristan Thrushes *Turdus eremita* (Ryan & Ronconi 2010).

The tiny population of Tristan Albatrosses have large chicks in the austral spring, and we counted all chicks. Atlantic Yellow-nosed Albatross nests containing eggs (or broken eggshells at newly refurbished nests) were counted in eight count areas, mostly centred around the top of the West Road. Together these areas accounted for around 20 % of the island's population (Ryan 2005). Sooty Albatrosses were harder to count, as most breed on the island's sheer cliffs, where their dark plumage makes them hard to detect. We counted incubating birds in the small colonies on the island plateau and thus accounted for perhaps 10 % of the island population (Ryan 2005). All albatross counts were conducted from late October to early November. Brown Skuas *Stercorarius antarcticus* breed throughout the island, with a major concentration around Skua Pond at West Point (Ryan & Moloney 2000). We recorded the locations of all nests encountered during October and November with a handheld GPS; given similar coverage of the island plateau, the counts in 1999, 2009, and 2018 are roughly comparable. Brown Noddies *Anous stolidus* and Antarctic Terns *Sterna vittata* were counted along the west coast from Warren's Cliff to Dirleton Point, with only incidental observations at the Waterfall during brief visits on 03 November and 01 December 2018. In addition, images were taken from a ship-based circumnavigation of the island on 01 December 2018. The east coast was not surveyed in 2009.

RESULTS

Spectacled Petrels

The burrow count in the top end of Ringeye Valley in 2018 (1 135) was down slightly from 2009 (1 210, compared to 970 in 2004 and 380 in 1999); some habitat was lost in this area due to the slumping and stripping of streambanks during a severe storm in winter 2017. During the systematic survey in 2018, we counted 2 922 Spectacled Petrel burrows in 25 transects (Fig. 1). The total length of transects sampled was 24.2 km, but the occupied range along transects (i.e., the distance from the first to the last burrow on each transect) was 18.7 km. Our counts were 2 449 burrows in 17.2 km in the main breeding range (21.2 km sampled), 75 burrows in 0.4 km at Denstone Hill (1.0 km sampled), and 398 burrows in 1.1 km at Round Hill (2.0 km sampled; Fig. 1). Average nest density in occupied habitat patches was 3.1 burrows per 100 m^2, but the density varied with habitat type. *Scirpus* terraces in bogfern heath supported a greater density of burrows (4.7 burrows per 100 m^2) than other habitats (Table 1). *Scirpus* terraces supported 62 % of burrows in 40 % of the occupied area, with most of the remaining burrows in bogfern heath (31 % of burrows in 43 % of the area) and *Spartina* tussock grassland (5 % of burrows in 12 % of the area, Table 1).

Assuming the five-metre limit on either side of the transect line was strictly followed, the number of burrows counted crudely extrapolated to some 39 200 burrows (Table 2). Based on the area of suitable habitat within the breeding range (bogfern heath 104 ha (1.04 km^2), *Scirpus* terraces 17 ha (0.17 km^2), and tussock grasslands 19 ha (0.19 km^2)), the extrapolated number of burrows is 47 316 (95 % confidence interval (CI) = 43 505–51 127); pooling bogfern heath and tussock grasslands gave a total of 58 471 burrows (49 453–67 490; Table 2). Burrow occupancy checks found that 8 % of perceived burrows were collapsed; in these cases, burrow entrances or fresh diggings did not lead to a nest chamber. However, five 'extra' burrows were found that were not detected in test areas, either through birds calling from hidden burrows or two burrows sharing a common entrance. Together, we suggest that some 97 % of counted burrows were potential breeding sites. Of these, 90 % contained petrels (range 84 % to 100 % at the five test areas), with petrels responding to playback in 91 % of occupied burrows. This gave an overall correction factor from burrow counts to occupied burrows of 87 %, which translated to a crude population estimate of 34 000 to 50 000 pairs (31 000 to 46 000 pairs in the main breeding range, 2 500 to 3 100 at Round Hill, and 500 to 950 at Denstone Hill), based on the three different extrapolation methods. The total population of Spectacled Petrels has increased by ~12 % per year since 1999, faster than the ~7 % per year growth rate estimated up to 1999 (Fig. 2). There is less confidence in the population estimates prior to 1999.

Northern Rockhopper Penguins

At a decadal scale, counts of penguins in beach parties at colonies have decreased on average by 1 % to 2 % per year since 1999 (Table 3). However, numbers were much lower in 2004 than in any other year (Table 3), despite similar numbers of birds breeding in both colonies accessible from Blenden Hall (Ryan 2005). This emphasises the crude nature of these counts as a tool to monitor population trends. Beach party counts also vary considerably both within days and seasonally through the breeding season (Ryan 2005). Counts from offshore tend to underestimate numbers of birds, even when using photographs; direct comparisons of beach parties at the three colonies around Salt Beach on 01 December gave ship-based counts of 975 birds (195, 560, and 220 for the Waterfall Slump, Salt Beach main, and Salt Beach north colonies, respectively), whereas shore-based counts made 2–3 h later averaged 50 % higher (285, 805, and 380 for the three colonies).

At a colony level, numbers of penguins at Where-the-Pig-Fell-Off decreased dramatically following a landslip that inundated part of the colony area between 2000 and 2004. A slip also covered much of the South Hill colony between 2011 and 2018. It is not known whether these slips happened during the breeding season. The decrease at South Hill has been offset to some extent by the formation of a new satellite colony west of the beach at Blenden Hall, some 160 m away from the main colony.

Estimates of nest densities at Warren's Cliff and Blenden Hall

TABLE 1
Numbers of Spectacled Petrel burrows counted and estimated densities (per 100 m^2) by habitat type, based on fixed line transects (10 m width) at Inaccessible Island, October–November 2018

Habitat	Patches	Length (m)	Burrows	Density	% Distance	% Burrows
Scirpus terraces	103	3 813	1 802	4.73	40.4	61.7
Bogfern	102	4 071	918	2.26	43.2	31.4
Spartina tussock	25	1 161	153	1.32	12.3	5.2
Riverbank	5	146	36	2.46	1.6	1.2
Ferns	3	109	7	0.64	1.2	0.2
Woodland	2	131	6	0.46	1.4	0.2
Totals	240	9 431	2 922			

TABLE 2
Estimates of the number of Spectacled Petrel burrows based on different extrapolation approaches using up to three habitat types: Bogfern heath (B), *Scirpus* terraces (T), and *Spartina* tussock grasslands (S)

Area	Extrapolation	Burrows (95 % CI)	Bogfern	Terrace	*Spartina*
Main range	crude area	35 755			
	B/T + S	53 679 (45 959–61 398)	49 502[a]		4 177
	B + T + S	42 576 (39 992–45 160)	29 640	8 760	4 177
Denstone Hill	crude area[b]	548			
	B/T + S	1 134 (869–1 398)	1 134[a]		0
	B + T + S	992 (701–1 282)	485	507	0
Round Hill	crude area[b]	2 905			
	B/T + S	3 659 (2 625–4 693)	3 659[a]		27
	B + T + S	3 748 (2 812–4 685)	2 821	927	27
Totals	crude area[b]	39 208			
	B/T + S	58 471 (49 453–67 490)	54 295[a]		4 177
	B + T + S	47 316 (43 505–51 127)	32 946	10 194	4 177

[a] Extrapolation for bogfern and *Scirpus* terraces combined
[b] The crude estimate simply corrects for the proportion of area not searched and so has no error term; habitat extrapolations use the standard error of the mean density estimate per habitat type to estimate the 95 % confidence intervals (CI) of the population per habitat.

did not differ significantly between 2009 and 2011, or between colonies (average overall was 1.48 nests per m^2, 95 % CI of the mean = 1.39–1.58, n = 82 quadrats). Colony areas decreased by ~10 % from 2009 to 2011 at both Warren's Cliff (3 395 to 3 060 m^2) and Blenden Hall (2 040 to 1 870 m^2).

Albatrosses

Tristan Albatrosses are biennial breeders and seldom, if ever, attempt to breed in successive years if they raise a chick (Ryan *et al.* 2001). As a result, the number of large chicks we counted in November reflects only a subset of the population and does not account for early nest failures. Since 1990, at most one chick has been counted each spring, and even visits in late summer (when adults are incubating) recorded only one incubating pair (Herian & Malan 2011, 2012). However, at least two pairs were present in 1999/2000, when there was one chick in late 1999 and another pair incubating in early 2000 (Ryan & Moloney 2000). In November 2018, two large chicks were present on the island: one on Gony Ridge at 37°18.357′S, 012°41.635′W and one on the ridge north of Gony Ridge at 37°18.226′S, 012°41.885′W. Numbers of Atlantic Yellow-nosed and Sooty Albatrosses were roughly constant from 2004 to 2009, then increased slightly from 2009 to 2018 (Tables 4 and 5).

Skuas, noddies, and terns

Reasonably comprehensive counts of Brown Skua pairs were obtained in 2009 and 2018 (Fig. 3). In 2004 there was little chance to explore the eastern plateau and only 66 pairs were counted (59 nests and 7 pairs holding territory). The northeastern coast between the Waterfall and Salt Beach was not visited in 2009 (two nests and one pair holding territory in 1999, one nest in 2018), and the southeastern coast from Pig Beach to Twin Falls was visited in 1999 only (one nest and one pair holding territory). However, there was better coverage on Harold's Plain in 2009 (nine nests) compared to 1999 (three nests and two pairs holding territory) or 2018 (two nests and two pairs holding territories). Despite these differences in coverage, there has been a steady increase in the number of pairs recorded either breeding or holding territory, from 18 in 1982 (Fraser *et al.* 1988) to 90 in 1999 (Ryan & Moloney 2000) to 107 in 2009 to 118 in 2018. This increase is also reflected in the well-studied Blenden Hall–West Point area (including Skua Pond), which supported 9 pairs in 1982, 10 in 1989, 21 in 1999, 22 in 2004, 28 in 2009, and 33 in 2018. Numbers of non-breeding birds attending the skua 'club' between Blenden Hall and West Point have remained roughly constant over this period (usually 50 to 90 birds).

Numbers of Brown Noddies nesting in trees in the vicinity of Blenden Hall also have increased steadily, from 4 pairs in the 1980s to 7 in 1999, 11 in 2004, 14 in 2009, and 19 in 2018 (Table 6). However, numbers breeding at the Waterfall appear to have decreased; we saw only a few pairs on the cliff here during a brief visit on 01 December 2018 (with none in the pine trees), compared to ~20 pairs in 1999 (Ryan & Moloney 2000) and perhaps 30 pairs in 1982 (Fraser *et al.* 1988). Noddy nests also were observed in rock crevices on the east side of South Hill as well as in *Phylica* trees along the river in middle Waterfall Valley and on the eastern slopes of Round Hill on the island plateau at around 300 m elevation.

Only two or three pairs of Antarctic Terns bred along the section of coast accessible from Blenden Hall (Warren's Cliff to Dirleton Point) in 2009 and 2018, with all nests at Warren's Cliff. Birds apparently no longer breed on the large rock exposure south of Skua Pond or at Dirleton Point (see Fraser *et al.* 1988). Elsewhere, they still breed on the low cliff behind the pine trees at the Waterfall (up to 10 pairs in 2018) and in rock crevices on the east side of South Hill (up to 20 pairs in 2018).

DISCUSSION

The populations of most monitored seabird species appear to be

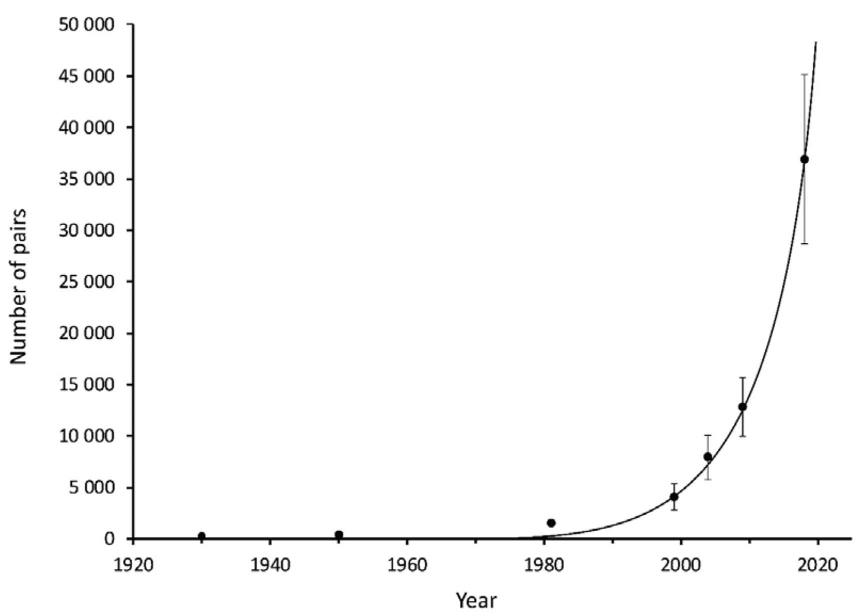

Fig. 2. Growth in the number of breeding pairs of Spectacled Petrels at Inaccessible Island. Error bars indicate standard error (no data for estimates made before 1999); the curve shows the best exponential growth model for surveys from 1999 to 2018 with an average annual growth rate of 11.4 % (R^2 = 0.997).

stable or increasing at Inaccessible Island, based on the most recent surveys in 2009 and 2018. The number of Northern Rockhopper Penguin may have decreased but counts of penguin beach parties give only a crude index of population size, and the apparent change in numbers was probably within the range of variance among and within years (considering the very low counts in 2004). Size estimates for the two colonies accessible from Blenden Hall decreased ~10 % from 2009 to 2011, but this might reflect inter-year variation rather than any long-term trend in the population. The fact that a new satellite colony has formed at Blenden Hall suggests that number of penguins is not decreasing dramatically, although it might include birds displaced by the landslip at the South Hill colony. More dedicated surveys of breeding colonies (e.g., Robson *et al.* 2011) are needed to assess whether the population is stable or decreasing. Considerable caution is needed to minimize disturbances (which potentially result in breeding failures) during such surveys.

The number of albatrosses has remained stable or perhaps increased slightly, despite all three species being listed as Endangered or Critically Endangered. The breeding population of Tristan Albatross at Inaccessible Island appears to be stable at around 2–3 pairs, with no indication of the population recovering to previous levels of several hundred pairs in the late 1870s (Fraser *et al.* 1988). The stable population indicates a low recruitment rate; juveniles may die at sea, recruit elsewhere, or perhaps die if they land among dense vegetation on the plateau of Inaccessible Island (Ryan *et al.* 1990). The only other breeding location is Gough Island, where depredation by introduced house mice *Mus musculus* is a major threat (Davies *et al.* 2015). A mouse eradication is planned for 2020.

Atlantic Yellow-nosed and Sooty Albatrosses breed on all three islands in the Tristan group and on Gough Island. Recent population trends on Tristan and Nightingale islands are not known, but at Gough Island, a small study colony of Atlantic Yellow-nosed Albatross has remained stable over a 30-year period (1982–2011) and a larger population that has been monitored since 2000 has also been stable (Cuthbert *et al.* 2014). The pattern is similar to the stable trend observed on Inaccessible Island since 2004. Similarly, monitored colonies of Sooty Albatrosses at Gough Island have been stable since 2000 (Cuthbert *et al.* 2014), as has been also true for Inaccessible Island (this paper). These encouraging findings could warrant a review of these species' global status, especially for Sooty Albatrosses, given their ongoing increase at Marion Island (Schoombie *et al.* 2016). However, climate change and the risk of introduced diseases are potential threats to the Tristan populations (see below).

Although numbers of Brown Noddies breeding on the west coast of Inaccessible Island have increased since the 1980s, there have been decreases of roughly the same order at other sites on the island. Antarctic Tern numbers appear to have decreased since the 1980s. The best data are for the west coast, where only 2–3 pairs bred at one site in 2018 compared to 16 pairs at three sites in 1982 (Fraser *et al.* 1988). There were also an estimated 70 pairs on the east coast in 1982, compared to about 10 pairs in 2018. The reason for this decrease is unclear, but it may be a consequence of climate warming, as Tristan is the northernmost breeding site for the species. By comparison, there has been a clear increase in the number of skuas breeding at Inaccessible Island. The counts from the Blenden Hall area are particularly telling, because this

TABLE 3
Trends in the numbers of Northern Rockhopper Penguins counted in beach parties during November at Inaccessible Island

Colony	1989[a]	1999[b]	2004	2009	2018
Blenden Hall	100	180	60	120	205
Blenden Hall new colony					55
Warren's Cliff	500	380	130	500	340
Where-the-Pig-Fell-Off	500	700	155	280	280
Salt Beach north	250	350	135	620	380
Salt Beach main	700	900	550	1 200	990
Waterfall Slump	150	180	65	85	215
Pig Beach	100	300	115	76	115
East Point	80	220	105	54	75
South Hill	nc[c]	130	nc[c]	100	25
Total	2 380	3 340	1 315	3 035	2 680

[a] Data from Ryan *et al.* 1990
[b] Data from Ryan & Moloney 2000
[c] nc = not counted

TABLE 4
Trends in the numbers of Atlantic Yellow-nosed Albatrosses in eight count zones on the plateau of Inaccessible Island

Count area	2004	2009	2018
Ringeye Valley	160	125	146
Slump	27	24	30
West Road valley	42	44	43
Cairn Peak South	28	29	32
Joe's River	43	44	42
Denstone River	56	58	78
Molly Bog	39	29	42
Total	359	353	413

TABLE 5
Trends in the numbers of Sooty Albatrosses in eight count zones on of Inaccessible Island

Count area	2004	2009	2018
West Road gulley	4	3	7
Slump scarp	6	6	8
Long Ridge South	14	11	12
Denstone Crag	5	9	6
2nd River West	2	10	8
Boulder Hill River	3	3	7
Gony Ridge	11	5	5
Dune Hills falls	6	1	8
Total	51	48	61

area is the best monitored and here the population has more than tripled since the 1980s. Skuas were killed and their nests destroyed on Tristan due to the perceived threat they pose to livestock, and similar persecution presumably occurred on Inaccessible Island up to the 1950s, when sheep were grazed on the island (Elliott 1957, Wace & Holdgate 1976). However, Tristan islanders seldom visited Inaccessible Island after the 1950s, allowing plenty of time for skua numbers to recover before the first population estimate in 1982 (Fraser *et al.* 1988). Skuas breeding at Inaccessible Island prey mainly on burrowing petrels (Ryan & Moloney 1991), and the distribution of nests has remained largely the same since 1999 (Fig. 3; Ryan & Moloney 2000). This suggests that petrel populations have remained relatively constant or possibly increased over the last 20 years.

Among monitored seabird species, the endemic Spectacled Petrel showed the greatest increase, continuing the species' long-term recovery following its near extinction due to depredation by introduced pigs *Sus scrofa* (Ryan 1998, Ryan & Moloney 2000, Ryan *et al.* 2006, Ryan & Ronconi 2011). The increase over the last few decades has been most obvious in peripheral breeding areas. For example, Spectacled Petrels apparently did not breed on Round Hill in the 1980s (Fraser *et al.* 1988), but there were some 120 burrows in 1999 (Ryan & Moloney 2000), 690 in 2009 (Ryan & Ronconi 2011), and by 2018 we counted 398 burrows by sampling along transects that covered ~1/7th of the available habitat. The total estimate was close to 3 000 burrows (Table 2). Range expansion was also detected on Denstone Hill, where isolated burrows were found on a ridge 180 m southwest of the 2009 range (transect 40.05; Fig. 1). Spectacled Petrels were found breeding for the first time in 2018 in mature *Phylica* woodland on the northern slopes of Denstone Hill. The ongoing range expansion presumably indicates a limited availability of favourable breeding sites within the core range, causing pairs to search for less crowded breeding sites elsewhere. This inference is supported by the high burrow-occupancy rates detected in all recent surveys (Ryan *et al.* 2006, Ryan & Ronconi 2011, this study). The population estimate from the new count technique has broader confidence intervals due to extrapolation from sampling less than 10 % of the species' range. However, by documenting exact ranges and burrow numbers per patch along repeatable transects, it will be easier to accurately detect future changes in both the density and distribution of Spectacled Petrel burrows. The latter procedure also reduces the amount of time necessary to conduct the survey.

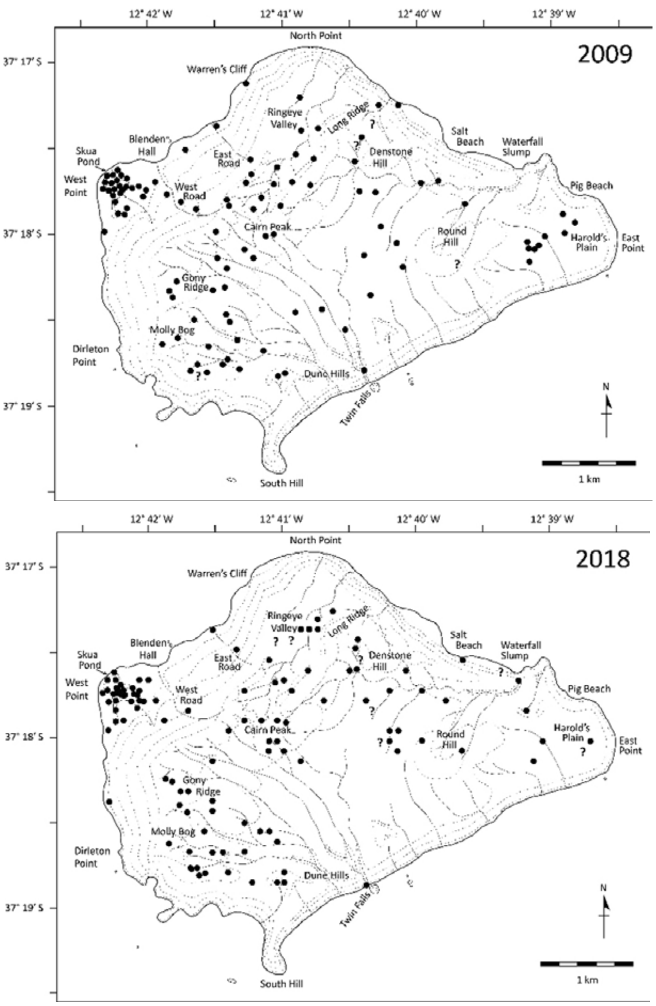

Fig. 3. The distribution of skua nests at Inaccessible Island in 2009 (n = 107) and 2018 (n = 118); "?" denotes pairs holding a territory but breeding was not confirmed.

Despite changing the count technique in 2018, the population growth of Spectacled Petrels has been consistently growing at ~12 % per year since the first systematic survey in 1999 (Fig. 2). This is faster than the annual growth rate up to 1998 of ~7 % per year, assuming that the estimates of ~50 pairs in 1937 (Hagen 1952), 200 in 1950 (Rowan *et al.* 1951), and 1 000 in 1982 (Fraser *et al.* 1988) are reasonably accurate. An increased growth rate over the last two decades might reflect reductions in the impact of fisheries bycatch on the species. The foraging ranges of Spectacled Petrels overlap substantially with long-line fisheries off Brazil and Uruguay (Bugoni *et al.* 2009, Reid *et al.* 2014), resulting in hundreds of Spectacled Petrels being killed in the 1980s and 1990s (Ryan *et al.* 2006 and references therein). Seabird bycatch rates in these fisheries have declined substantially thanks to the implementation of various mitigation methods (e.g. bird-scaring lines, branch line weighting, and night-time setting of hooks; Bugoni *et al.* 2008b, Jiménez *et al.* 2010, Melvin *et al.* 2013, Jiménez *et al.* 2019, Santos *et al.* 2019), with decreases in incidental capture detected from around the time of the first systematic census in 1999 (Ryan *et al.* 2006 and references therein). It is particularly encouraging that the number of seabirds killed has decreased while the population of Spectacled Petrels has increased, suggesting that the benefits

TABLE 6
Long-term trends in the numbers of Brown Noddy nests in trees around Blenden Hall, Inaccessible Island

Count area	1980s[a]	1999[b]	2004	2009	2018
Hut/Nelson's Gulch	2	0	0	2	3
First Apples	1	4	5	5	4
Second Apples	0	2	2	3	3
Wilkins' Copse	1	0	4	4	7
East Road	nc[c]	1	0	1	2
Total	4	7	11	15	19

[a] Data from Fraser *et al.* 1988 & Ryan *et al.* 1990
[b] Data from Ryan & Moloney 2000
[c] nc = not counted

of these mitigation measures are even greater for this species than is apparent from the changes in bycatch rate alone. Fortunately, Spectacled Petrels appear to be less susceptible to capture on long-lines than their close relative, the White-chinned Petrel *Procellaria aequinoctialis*, which is the species most often killed on long-lines in the Southern Ocean (Bugoni *et al.* 2008a, Jiménez *et al.* 2009, Rollinson *et al.* 2017, Jiménez *et al.* 2019).

The Spectacled Petrel provides an excellent example of how a petrel population can recover following near extinction due to depredation by an introduced mammal. Such success stories are important for promoting further restoration programmes that involve removing introduced predators from seabird breeding islands (Brooke *et al.* 2018). The IUCN down-listed the Spectacled Petrel from Critically Endangered to Vulnerable in 2007, and, although its breeding population continues to increase, it still qualifies as globally Vulnerable under criterion D2 ("population very small or restricted") since the entire breeding population is confined to a single 14 km^2 island and has a total breeding range of less than 5 km^2. It remains susceptible to fishery impacts, given that Spectacled Petrels probably are killed by high-seas tuna fisheries, for which there are scant data on seabird bycatch rates (Reid *et al.* 2014). However, threats at the colony pose the greatest concern, despite Inaccessible Island's status as a nature reserve and World Heritage Site.

Seabirds are at risk from peat slips, which kill breeding birds (e.g., Ryan 1993) and reduce suitable habitat for burrow-nesting birds (e.g., loss of deep soils on favourable slopes), penguins (e.g., when coastal breeding sites are destroyed by a deluge of debris from slips), and albatrosses (e.g., nest mounds and surrounding substrate washed away). The severe winter storm that washed away roads and buried pastures on Tristan in 2017 also caused extensive peat slips on Nightingale Island (BJD pers. obs.) and was probably responsible for the many recent slips observed on the plateau of Inaccessible Island in 2018. These slips contributed to the apparent slight decrease in Spectacled Petrels breeding in the top of Ringeye Valley. Ongoing climate change is likely to increase the frequency and intensity of severe weather events, with possibly adverse impacts for Spectacled Petrels and other seabirds breeding on Inaccessible Island. However, introduced diseases from domestic poultry pose perhaps the most serious threat. Jaeger *et al.* (2018) document the severe demographic impact of avian cholera on albatrosses and penguins breeding on Amsterdam Island. The disease apparently was introduced from poultry kept on the island (Jaeger *et al.* 2018). Skuas provide a potential vector for transferring poultry diseases from the main island of Tristan da Cunha to Inaccessible Island (Jaeger *et al.* 2018, Cerdà-Cuéllar *et al.* 2019). Tristan's biosecurity measures need to be particularly stringent on any poultry imports to reduce the risk of diseases such as avian cholera, avian influenza, or Newcastle disease from reaching Tristan.

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