



Review

The conservation status and priorities for albatrosses and large petrels



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ABSTRACT

Seabirds are amongst the most globally-threatened of all groups of birds, and conservation issues specific to albatrosses (Diomedidae) and large petrels (*Procellaria* spp. and giant petrels *Macronectes* spp.) led to drafting of the multi-lateral Agreement on the Conservation of Albatrosses and Petrels (ACAP). Here we review the taxonomy, breeding and foraging distributions, population status and trends, threats and priorities for the 29 species covered by ACAP. Nineteen (66%) are listed as threatened by IUCN, and 11 (38%) are declining. Most have extensive at-sea distributions, and the greatest threat is incidental mortality (bycatch) in industrial pelagic or demersal longline, trawl or artisanal fisheries, often in both national and international waters. Mitigation measures are available that reduce bycatch in most types of fisheries, but some management bodies are yet to make these mandatory, levels of implementation and monitoring of compliance are often inadequate, and there are insufficient observer programmes collecting robust data on bycatch rates. Intentional take, pollution (including plastic ingestion), and threats at colonies affect fewer species than bycatch; however, the impacts of disease (mainly avian cholera) and of predation by introduced species, including feral cats (*Felis catus*), rats (*Rattus* spp.) and house mice (*Mus musculus*), are severe for some breeding populations. Although major progress has been made in recent years in reducing bycatch rates and in controlling or eradicating pests at breeding sites, unless conservation efforts are intensified, the future prospects of many species of albatrosses and large petrels will remain bleak.

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Contents

1. Introduction	170
2. Taxonomy	170
3. Geographic distribution	171
3.1. Breeding sites	171
3.2. At-sea distribution	171
4. Population status and trends	173
5. Threats: bycatch in fisheries	174
5.1. Scale, contributing factors and impacts	174
5.2. Progress in mitigating threats from fisheries	175
6. Threats: intentional take or killing at sea	175
7. Threats: pollution, debris and discarded fishing gear	176
8. Threats: alien species at breeding sites	176

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8.1. Impacts of alien species	176
8.2. Progress in managing alien species	177
9. Threats: pathogens	177
10. Threats: climate change.	177
11. Threat prioritisation	179
12. Future challenges for albatross and petrel conservation	179
Acknowledgements	180
Appendix. Supplementary data	180
References.	180

1. Introduction

According to the IUCN Red List criteria, which relate to population size, trends, and the extent and fragmentation of breeding distributions, seabirds are amongst the most threatened of all groups of birds (Croxall et al., 2012). Albatrosses and petrels are long-lived, have high adult survival rates, delayed sexual maturity and low fecundity; all lay single-egg clutches, and nine species (all of which are albatrosses) breed biennially if successful in raising a chick (Warham, 1990). Given these extreme life-history attributes, changes in adult mortality have a much greater impact on population trajectories than variation in other demographic parameters, including breeding success, proportion of deferring breeders, juvenile survival and recruitment (Arnold et al., 2006; Croxall and Rothery, 1991; Moloney et al., 1994; Véran et al., 2007). All species have wide at-sea distribution during the breeding and non-breeding seasons; these extensive foraging ranges overlap with, and so put them at potential risk from multiple fisheries in national and international waters (Baker et al., 2007; Delord et al., 2010; Phillips et al., 2006).

Incidental mortality of seabirds in fisheries (hereafter “bycatch”), particularly of albatrosses and petrels, became a major conservation concern in the late 1980s (Brothers, 1991; Murray et al., 1993; Weimerskirch and Jouventin, 1987). Initial evidence came from numerous recoveries in longline fisheries of wandering albatrosses (*Diomedea exulans*) ringed at South Georgia (Islas Georgias del Sur) (Croxall and Prince, 1990), and estimates of very high bycatch from the Japanese tuna fishery off Australia (Brothers, 1991). Although based on very small samples, the inferred mortality coincided with declines in albatross populations in the sub-Antarctic, and so it was strongly suspected that fisheries bycatch was a critical factor (Croxall and Prince, 1990; Prince et al., 1994b; Weimerskirch and Jouventin, 1987). High rates of seabird bycatch were subsequently confirmed in a wide range of long-line fisheries (Brothers et al., 1999b; Gales, 1998; Tasker et al., 2000). Although attention focused initially on industrial longlining, bycatch by trawl and artisanal fleets have also been identified as major sources of mortality for many albatrosses and petrels (Croxall et al., 2012; Favero et al., 2010; Maree et al., 2014; Sullivan et al., 2006b).

Solving a conservation problem as pervasive as bycatch for species as wide-ranging as albatrosses and large petrels requires concerted management actions that cover both national and international waters. This motivated the development of the Agreement on the Conservation of Albatrosses and Petrels (ACAP) as a daughter agreement of the Convention on Migratory Species (Bonn Convention), and its ratification in 2004 (Cooper et al., 2006). Although bycatch remains the main threat to many species and hence the contributing factors and demographic consequences are principal foci in this review, albatrosses and petrels also face a range of other threats on land and at sea, including impacts of invasive species, degradation or loss of nesting habitat, disease, pollution and climate change (see below). Consequently, the Action Plan of ACAP addresses topics that include habitat conservation and restoration, management of human activities, research and monitoring, education and public awareness, collation of information and implementation (Agreement on the Conservation of Albatrosses and Petrels, 2001; Cooper et al., 2006). The purpose of this paper is to review the

taxonomy, breeding and at-sea distributions, population status and trends, and marine and terrestrial threats to the 22 albatrosses and seven large petrels (*Macronectes* and *Procellaria* spp.) listed under ACAP, and report recent progress in addressing those threats and the priority conservation actions for the future. In order to maintain taxonomic and geographic coherence, the review does not cover the two species of shearwater added to the ACAP list since 2009 (Balearic shearwater *Puffinus mauretanicus* and pink-footed shearwater *Puffinus creatopus*). Unless indicated otherwise by a supporting citation, data in tables and figures reflect published and unpublished data submitted to the ACAP database, available at www.acap.aq.

2. Taxonomy

Although >80 albatross taxa have been formally described since the mid 1700s (Robertson and Nunn, 1998), many were based on specimens collected at sea from unknown breeding locations and later revealed to be age-related plumage morphs of previously-described species. Taxonomic confusion was compounded by a scarcity of information on breeding behaviour and distribution, strong natal philopatry which precluded recognition of genuine physiological or behavioural barriers to gene flow (because contact between individuals from disparate populations is rare), and unusually low levels of genetic divergence even between what appear to be very different species (Nunn et al., 1996; Nunn and Stanley, 1998). This reduces the power of genetic studies to delineate species boundaries (Burg and Croxall, 2001, 2004; Double et al., 2003).

The taxonomic debate surrounding albatrosses was revisited when a new taxonomy was proposed by Robertson and Nunn (1998). This largely applied the Phylogenetic Species Concept and recognised 24 albatross species; however, some decisions were controversial (Penhallurick, 2012; Penhallurick and Wink, 2004; Rheindt and Austin, 2005). Although the recommendation to re-establish four genera (resurrecting *Phoebastria* and *Thalassarche*) has been universally accepted, there is no current consensus at the species level; subsequent taxonomic treatises, field guides and reviews recognised between 13 and 24 albatross species (e.g. Brooke, 2004; Chambers et al., 2009; Christidis and Boles, 2008; Onley and Scofield, 2007; Penhallurick and Wink, 2004; Shirihai, 2002). Acknowledging that taxonomic confusion could hamper conservation, ACAP established a Taxonomy Working Group with a remit to develop a defensible species list based upon peer-reviewed literature and a transparent decision-making process. This group largely follows guidelines in Helbig et al. (2002) which apply a relaxed version of the General Lineage Species Concept, focusing on diagnostic characteristics and evidence for distinct evolutionary trajectories. After assessing the splits advocated by Robertson and Nunn (1998), the conclusion was that two (Pacific albatross *Thalassarche bulleri platei* and Gibson's albatross *Diomedea antipodensis gibsoni*) of the 24 terminal albatross taxa could not be justified as separate species based on available data. The recognition of 22 albatross species by ACAP was later endorsed by Birdlife International (2015), the official IUCN Red List Authority.

Most regional or global taxonomic authorities now recognise 21 or 22 albatross species, depending on whether shy (*Thalassarche cauta*)

and white-capped albatross (*T. steadi*) are considered — which they are by ACAP — to be separate species (BirdLife International, 2015; Gill and Donsker, 2016; Tennyson, 2010). The argument by a minority for a return to 13 or 14 albatross species is based largely around percentage sequence divergence (Christidis and Boles, 2008; Penhallurick and Wink, 2004). Unsurprisingly, the sequence divergence between sister taxa in the 14-species taxonomy is greater than for the 22-species taxonomy; indeed, divergence is very low between many sister taxa in the latter (<1% cytochrome b, Chambers et al., 2009; Nunn et al., 1996; Nunn and Stanley, 1998). However, this alone should not preclude recognition at the species level because neutral mitochondrial markers are insensitive to rapid radiations (Chambers et al., 2009; Rheindt and Austin, 2005). Moreover, no one level of sequence divergence can define a species event; this is particular pertinent for albatrosses, as molecular evolution is highly variable within the Procellariiformes and larger species show slower rates (Nunn and Stanley, 1998).

The other taxonomic dispute concerns northern (*Macronectes halli*) and southern giant (*Macronectes giganteus*) petrels, which are morphologically similar and show low sequence divergence (Nunn and Stanley, 1998; Penhallurick and Wink, 2004). However, a rare white plumage phase only occurs in the southern giant petrel, and this species has a different bill tip colour and in areas of sympatry breeds about 6 weeks later than its congener (Bourne and Warham, 1966; Brown et al., 2015). Few now argue against separate species status (but see Penhallurick and Wink, 2004). Finally, spectacled petrel (*Procellaria conspicillata*) was at one time considered to be a subspecies of white-chinned petrel (*Procellaria aequinoctialis*), but has since been accorded species status, reflecting vocal, plumage, structural and genetic differences (Ryan, 1998; Techow et al., 2009).

3. Geographic distribution

3.1. Breeding sites

The global breeding distributions of the albatrosses and large petrels vary greatly in geographic extent. Breeding sites, as listed by ACAP, are usually an entire, distinct island or islet, or rarely, section of a large island (>3000 km²), and each species-site combination is included separately, i.e., two species breeding in the same area constitute two breeding sites. If the few sites with tiny populations (<10 breeding pairs) are excluded, five albatrosses (wandering, grey-headed *Thalassarche chrysostoma*, black-browed *Thalassarche melanophris*, sooty *Phoebastria fusca* and light-mantled *Phoebastria palpebrata* albatrosses), the two giant petrels, and two of the *Procellaria* petrels (white-chinned and grey petrels *Procellaria cinerea*) have a circumpolar breeding distribution, with populations in every Southern Ocean basin; eight albatrosses (Antipodean *Diomedea antipodensis*, Buller's *Thalassarche bulleri*, Campbell *Thalassarche impavida*, Chatham *Thalassarche eremita*, white-capped, northern royal *Diomedea sanfordi*, southern royal *Diomedea epomophora* and Salvin's albatrosses *Thalassarche salvini*) and two *Procellaria* petrels (Westland *Procellaria westlandica* and black *Procellaria parkinsoni* petrels) breed only around New Zealand; two albatrosses (Tristan *Diomedea dabbenena* and Atlantic yellow-nosed albatrosses *Thalassarche chlororhynchos*), and spectacled petrel breed only on islands in the Atlantic Ocean; two albatrosses (Indian yellow-nosed *Thalassarche carteri* and Amsterdam albatrosses *Diomedea amsterdamensis*) only in the Indian Ocean; three albatrosses (Laysan *Phoebastria immutabilis*, black-footed *Phoebastria nigripes* and short-tailed albatrosses *Phoebastria albatrus*) only in the North Pacific; shy albatross only in Tasmania, and; waved albatross *Phoebastria irrorata* only regularly in the Galápagos islands (Fig. 1). Seven albatross and three *Procellaria* petrel species are endemic to a single island or island group (Fig. 1). Almost all breeding colonies are on remote islands, ranging in size from tiny rocky islets to Grande Terre, Kerguelen Islands (6675 km²) and the South Island, New Zealand.

The ACAP database includes virtually all the existing census data for the 29 species in this review, and allows the identification of internationally important breeding sites — single islands or, in a few cases, peninsulas or small island groups — that hold >1% of the global population (Appendix A). Using this definition, and bearing in mind the caveats that there are no census data for around 22% of breeding sites (particularly those of the burrow-nesting *Procellaria* petrels and light-mantled albatross), and some counts are of low reliability or more than a decade old, most albatrosses and larger petrels breed at relatively few sites; for 16 of the 29 species, there are only 1–3 sites with >1% of global numbers. Only for a minority of albatrosses (8 of 22 species) are there ≥5 breeding sites with >1% of the global population, and only for the five albatrosses and the two giant petrels that have circumpolar breeding distributions (see above) are there ≥9 sites that hold >1% of global numbers. No species breeds at ≥3 sites that each hold >10% of the global population. The restricted breeding distribution of many species increases their vulnerability to localised threats (see below), and is reflected in the assignment by IUCN of some albatrosses and *Procellaria* petrels to a threat category of Vulnerable even though the global populations are not thought to be decreasing (see below).

3.2. At-sea distribution

Albatrosses and large petrels are exceptionally wide-ranging, frequently travelling 100s to 1000s of km on a single foraging trip that can extend to a straight-line distance of >2000 km from the colony (Peron et al., 2010b; Phillips et al., 2004; Weimerskirch et al., 1993). This reflects trip durations during incubation and chick-rearing that can be of 2–3 weeks, although it is more common for the adult to return and feed its chick after 2–4 days, especially during brood-guard (Phillips et al., 2005a; Torres et al., 2013). As the degree of central-place foraging constraint varies with breeding phase, so too does the extent of at-sea distributions; this is sometimes associated with a change in habitat use from oceanic, distant shelf or shelf-slope regions in the pre-laying and incubation periods, to neritic waters much closer to the colony in brood-guard, and then a return to more distant waters for the remainder of chick-rearing (Phillips et al., 2006; Wakefield et al., 2011; Weimerskirch et al., 1993). During chick-rearing, parents may adopt a dual foraging strategy, involving the alternation of long and short trips as they balance the demands of chick provisioning with self-maintenance (Weimerskirch et al., 1994).

Almost all the albatross and large petrel species have been tracked at some stage while breeding, and many during the nonbreeding season (although only in recent years and many data are unpublished), whereas there are relatively few tracks from juveniles and immatures (De Grissac et al., 2016; Dias et al., 2014). During the nonbreeding period, many species make a directed, long-distance migration to a productive upwelling, shelf or frontal system, sometimes in a different ocean basin, and return to the colony can involve a circumnavigation of the Antarctic continent (Croxall et al., 2005). There are, however, numerous exceptions and contrasting strategies. Thus, Atlantic yellow-nosed albatrosses from Tristan da Cunha and Gough, and black-browed albatrosses from South Georgia migrate a few thousand km east across the south Atlantic Ocean to the Benguela Upwelling system, where they overlap with nonbreeding white-chinned petrels from colonies in the Indian Ocean, and some white-capped and shy albatrosses that have travelled much longer distances west from the Auckland Islands and Tasmania, respectively (Peron et al., 2010b; Phillips et al., 2005b). In contrast, white-chinned petrels, also from South Georgia, migrate only to the Patagonian Shelf or the Humboldt Upwelling; in the former, they overlap with wintering black-browed albatrosses from the Falklands (Malvinas) and northern royal albatrosses from New Zealand, and in the latter with several species of albatrosses and large petrels from New Zealand, including Salvin's, Buller's, Chatham and Antipodean albatrosses, black and Westland petrels (Landers et al., 2011; Nicholls et al., 2002; Phillips et al., 2006; Spear et al., 2003; Walker and Elliott, 2006). Even within the

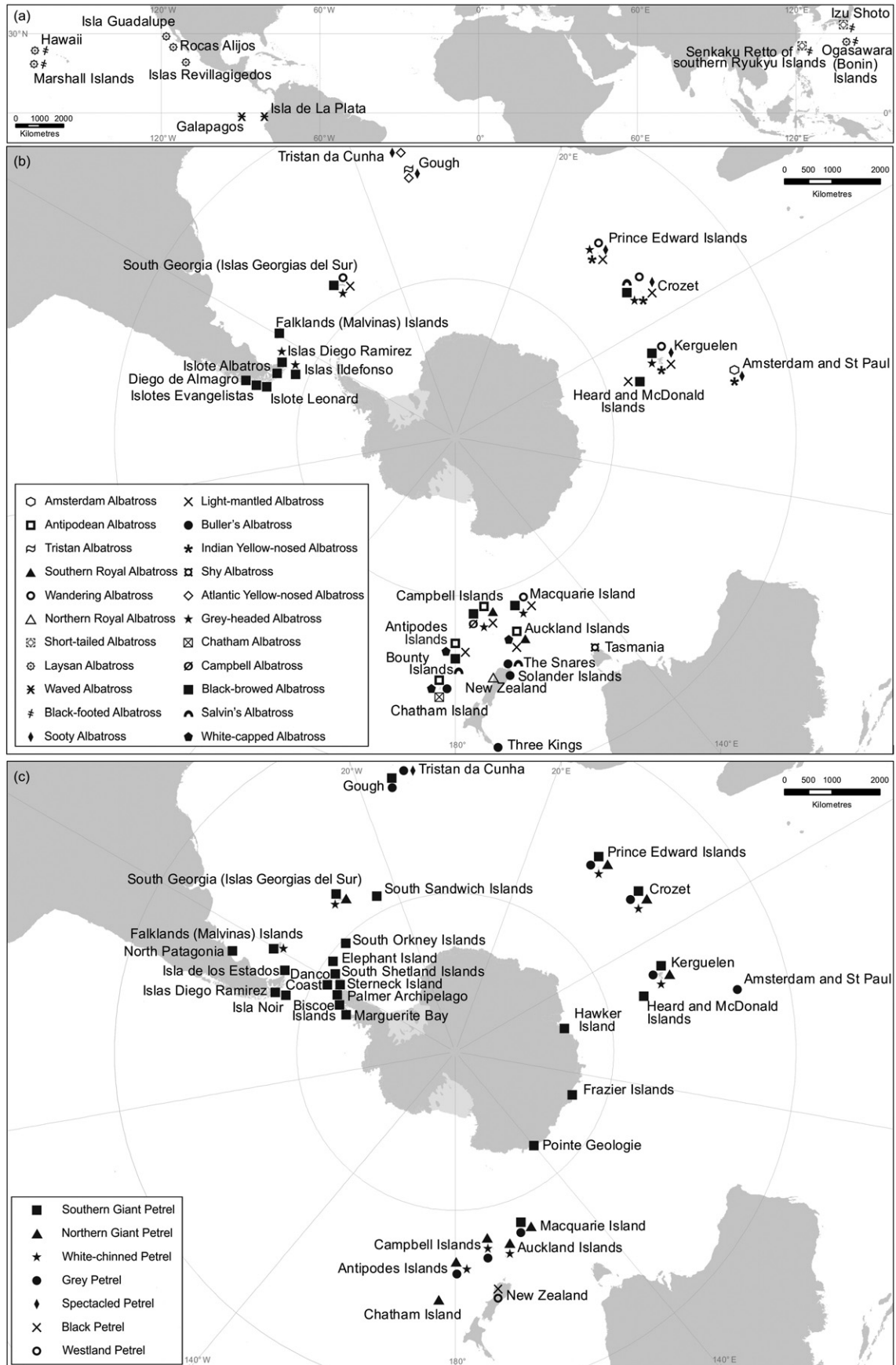


Fig. 1. Breeding locations of (a) albatrosses in equatorial and north Pacific Ocean, (b) albatrosses in the Southern Ocean, and (c) *Macronectes* and *Procellaria* petrels in the Southern Ocean.

same population, there is often extensive variation among individuals in movements and distribution (Croxall et al., 2005; Phillips et al., 2006; Phillips et al., 2005b).

Albatrosses and large petrels display diverse habitat preferences, reflecting the broad range of oceanographic conditions in waters around their scattered colonies and in the more distant regions used at other times of year. They can be specialists or generalists, reflected in the proportion of time spent utilising tropical, subtropical, sub-polar or polar, and continental shelf, island shelf, shelf-slope or oceanic waters at different times of year (Peron et al., 2010b; Phillips et al., 2006; Phillips et al., 2005b; Walker and Elliott, 2006). Several species exhibit pronounced sexual segregation, with females tending to feed at lower latitudes or further from colonies than males, attributed to competition between sexes or habitat specialisation, and related in some, but not all species, to sexual size dimorphism (Bartle, 1990; González-Solís et al., 2000; Weimerskirch et al., 1993). There can also be partial or complete spatial segregation between juveniles and adults (Alderman et al., 2010; Gutowsky et al., 2014; Weimerskirch et al., 2014). Even in areas of spatial overlap, species usually differ in at-sea activity patterns (e.g. frequency of landings, flight and resting bout durations), reflecting the distribution of preferred prey or degree of nocturnality, among others (Mackley et al., 2010; Phalan et al., 2007; Weimerskirch and Guionnet, 2002). There are also large differences in diving capability; albatrosses and, given anatomical similarities, probably giant petrels, are much poorer divers than *Procellaria* petrels (Hedd et al., 1997; Prince et al., 1994a; Rollinson et al., 2014). Intra- and inter-specific variation in distribution, habitat preferences, dive depth and other aspects of behaviour have major implications for the degree of overlap and hence risk of bycatch in different fisheries (see below).

4. Population status and trends

The 29 species of albatrosses and large petrels (*Macronectes* and *Procellaria*) included here collectively comprise almost 3 million pairs breeding at 571 sites, across multiple jurisdictions. Trends vary between sites and species, but globally, over the 2 decades from 1993 to 2013,

about 38% of these species declined, 28% increased, 28% were stable, and the trend for 7% (2 species) was unknown. Nineteen species (66%) are considered to be threatened (Vulnerable, Endangered or Critically Endangered) by IUCN (Fig. 2, Table 1). Three species qualify as Critically Endangered, all with very restricted breeding ranges. Two are declining: the Tristan albatross because of a combination of bycatch and predation of chicks by introduced house mice *Mus musculus* (Wanless et al., 2009), and the waved albatross because of bycatch and intentional take for human consumption (Anderson et al., 2008). The Amsterdam albatross is increasing as it recovers from degradation of its nesting habitat and impacts of longline fisheries (Inchausti and Weimerskirch, 2001), but remains in perilously low numbers (31 breeding pairs; Table 1). A further five albatross species are Endangered; grey-headed and Indian yellow-nosed albatrosses because of rapid population decline at South Georgia and Amsterdam Island, respectively; sooty albatross seems to be declining based on limited data; Atlantic yellow-nosed albatross appears to be stable, but with low confidence in the trend data, and; the current trend for northern royal albatrosses is uncertain.

Eleven species (seven albatrosses and four *Procellaria* petrels) are Vulnerable; in some cases, this reflects restricted breeding range and not a declining population (Fig. 2, Table 1; www.iucn.org). Eight of these species breed within the jurisdiction of one country, seven in New Zealand. The populations of four species (Chatham, Campbell, and southern royal albatrosses, and Westland petrel) are considered stable. Wandering and Antipodean albatross, and black petrel are in decline. Although the short-tailed albatross is recovering rapidly from near-extinction because of careful management, the population remains at <650 breeding pairs each year (Finkelstein et al., 2010, Table 1). By comparison, the white-chinned petrel is far more abundant (c.1 million breeding pairs) but the limited trend data suggest a steep decline of the largest population (South Georgia) from the 1980s to the later 1990s, as a result of incidental mortality in fisheries (Phillips et al., 2006). Eight species are Near Threatened, two of which are increasing, the black-browed and black-footed albatrosses (Fig. 2, Table 1). Limited trend data are available for light-mantled albatross (probably stable), white-

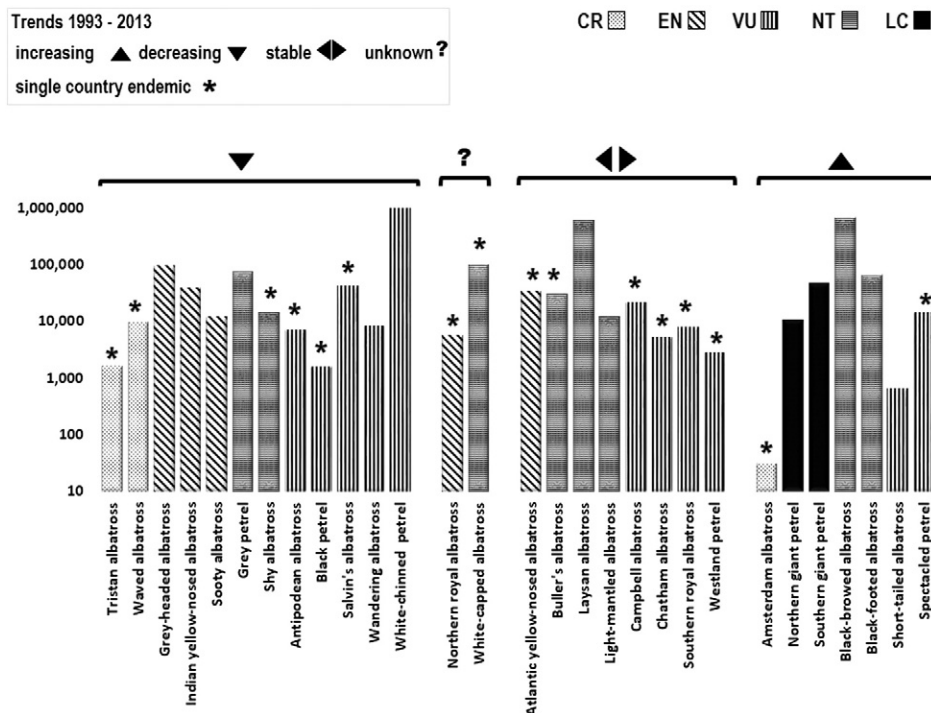


Fig. 2. Annual breeding population size, IUCN status and population trend (1993–2013) of albatrosses and large petrels (*Macronectes* and *Procellaria* spp.) IUCN status: CR = critically endangered, EN = endangered, VU = vulnerable, NT = near threatened, LC = least concern. <www.iucnredlist.org>.

Table 1
Summary of status, trends, number of breeding sites and recent population estimate for albatrosses and large petrels (*Macronectes* and *Procellaria* spp.).

Species	No. sites ^a	Single country endemic	Breeding freq. ^b	Annual breeding pairs	Latest census year by site	Current trend 1993–2013 ^c	Trend confidence	IUCN status (2015 ^d)
Amsterdam albatross	1	France	B	31	2013	↑	High	CR
Tristan albatross	1	UK	B	1650	2014	↓	High	CR
Waved albatross	3	Ecuador	A	9615	2001–2013	↓	Low	CR
Atlantic yellow-nosed albatross	6	UK	A	33,650	1974–2011	↔	Low	EN
Grey-headed albatross	29		B	98,084	1982–2015	↓	Medium	EN
Indian yellow-nosed albatross	6		A	39,319	1984–2009	↓	Medium	EN
Northern royal albatross	5	NZ	B	5782	1995–2013	?	–	EN
Sooty albatross	15		B	12,103	1974–2014	↓	Very low	EN
Antipodean albatross	6	NZ	B	7029	1995–2013	↓	Medium	VU
Black petrel	2	NZ	A	1577	1998–2014	↓	Medium	VU
Campbell albatross	2	NZ	A	21,648	2012	↔	Low	VU
Chatham albatross	1	NZ	A	5245	2011	↔	Medium	VU
Salvin's albatross	12	NZ	A	41,111	1986–2013	↓	Low	VU
Short-tailed albatross	2		A	661	2002–2014	↑	High	VU
Southern royal albatross	4	NZ	B	7924	1989–2014	↔	Medium	VU
Spectacled petrel	1	UK	A	14,400	2010	↑	High	VU
Wandering albatross	35		B	8359	1981–2015	↓	High	VU
Westland petrel	1	NZ	A	2827	2011	↔	Low	VU
White-chinned petrel	74		A	1,160,152	1984–2013	↓	Very low	VU
Black-browed albatross	65		A	691,046	1982–2015	↑	High	NT
Black-footed albatross	15		A	66,376	1995–2014	↑	High	NT
Buller's albatross	10	NZ	A	30,069	1971–2014	↔	Low	NT
Grey petrel	17		A	75,610	1981–2012	↓	Very low	NT
Laysan albatross	17		A	610,496	1982–2014	↔	High	NT
Light-mantled albatross	71		B	12,082	1954–2014	↔	Very low	NT
Shy albatross	3	Australia	A	14,353	2015	↓	Low	NT
White-capped albatross	5	NZ	?	100,525	1995–2013	?	–	NT
Northern giant petrel	50		A	10,594	1973–2014	↑	Medium	LC
Southern giant petrel	119		A	47,516	1958–2015	↑	Medium	LC

^a Site: usually an entire, distinct island or islet, or rarely, section of a large island (>3000 km²). Each species-site combination is considered separately, i.e., two species breeding in the same area constitute two breeding sites ACAP database. <<https://data.acap.aq>>. 14 July 2015.

^b Breeding frequency: A = annual, B = biennial.

^c Trend: ↑ increasing, ↓ declining, ↔ stable, ? unknown.

^d IUCN status: CR = critically endangered, EN = endangered, VU = vulnerable, NT = near threatened, LC = least concern. <www.iucnredlist.org>.

capped albatross (trend uncertain), and grey petrel and shy albatross (declining). Laysan and Buller's albatrosses are stable. The two species of Least Concern are the northern and southern giant petrels, both of which are increasing.

There are no counts within the last decade for 64 breeding sites (of 12 albatross and four petrel species) that were known to hold >1% of the global population (Table 1), or for any site in 13 island groups (of 5 albatross and three petrel species) that together held >1% of the global population (Appendix B). In addition, the Prince Edward Islands potentially hold > 1% of global numbers of grey petrels, but no estimate is available. Adult and juvenile survival rates, and breeding success are known from at least one site for all species except for adult survival (spectacled petrel), juvenile survival (Chatham, Salvin's, southern royal, white-capped, light-mantled and short-tailed albatrosses, and northern giant and spectacled petrel), and breeding success (Chatham and Salvin's albatrosses, and spectacled petrel), although in some cases, data have been collected but not published. Data gaps often reflect the logistical challenges of working at remote islands, and funding limitations given the large number of breeding sites in some jurisdictions.

5. Threats: bycatch in fisheries

5.1. Scale, contributing factors and impacts

Bycatch of seabirds in longline fisheries occurs when birds attack baited hooks and become hooked and drowned as the line sinks (Brothers, 1991). In trawl fisheries, birds foraging on discards or offal

(hereafter “discards”) may be injured or killed on collision with net-monitoring and warp cables, dragged underwater and drowned when their wings become entangled around the warp, or become entangled in nets (Sullivan et al., 2006a; Watkins et al., 2008). Incidental capture in gillnet fisheries is due mostly to entanglement while diving for prey (Melvin et al., 1999; Waugh et al., 2011).

Bycatch is often unevenly distributed; biases can be towards males or females, adults or immatures, and depend on fishing area, gear type or season (Bugoni et al., 2011; Delord et al., 2005; Gales et al., 1998). Variation in the sex and age classes most at risk are often due to differences in foraging distributions at each stage of the annual (breeding and nonbreeding) cycle, and hence the relative overlap with high-risk fisheries (Alderman et al., 2011; Baker et al., 2007; Cuthbert et al., 2005; Delord et al., 2010). Bycatch rates of birds in different life-history stages have implications for demography and population trajectories, including time lags before detection and potential recovery (Dillingham and Fletcher, 2011).

Although the volume and reliability of bycatch information are still severely limited for many areas and fisheries, particularly artisanal and gillnet, there has been a general improvement in the last decade, with better sampling coverage (Anderson et al., 2011; Richard and Abraham, 2014; Żydelski et al., 2013). Regardless, the scale of bycatch is huge. An assessment for longline fisheries just in the Atlantic Ocean estimated c.48,500 seabirds were killed in 2003–2006 (Klaer, 2012; Tuck et al., 2011). In the most recent estimate at the global level, >160,000, and potentially >320,000 seabirds are killed annually in longline fisheries, a large portion of which are albatrosses and large petrels (Anderson et al., 2011). Estimated annual global bycatch in gillnet

fisheries is even higher, and although only a small proportion are procellariids, the impact on species such as the waved albatross may be severe (Żydelis et al., 2013).

Many operational, environmental and ecological factors influence the nature and extent of seabird bycatch (Gómez Laich et al., 2006; Klaer and Polacheck, 1998). Albatrosses and larger petrels are particularly susceptible; they scavenge on food items near the sea surface, have a propensity to follow vessels, and possess large gapes so can ingest baited hooks (Brothers et al., 2010; Brothers et al., 1999a). They also have a competitive advantage over smaller birds when attempting to access bait and discards (Brothers, 1991; Jimenez et al., 2011), although there are differences in feeding behaviour and vulnerability to capture among species of similar size (Brothers et al., 2010). The *Procellaria* petrels are more proficient divers, as are shearwaters *Ardenna* species, and in multi-species feeding assemblages can seize baited hooks at depths below those accessible to larger species; by returning those to the surface, bycatch of albatrosses is increased (Jimenez et al., 2012a). Bycatch in trawl fisheries is similarly influenced by species-specific differences in size and manoeuvrability; the large albatrosses are particularly susceptible to injury on warp cables (Favero et al., 2010; Sullivan et al., 2006a; Sullivan et al., 2006b; Watkins et al., 2008).

Although bycatch is now recognised as the most pervasive threat for albatrosses and large petrels, there are populations (spectacled petrel, and white-chinned petrels at Marion Island) which are increasing following the removal of terrestrial threats, despite ongoing mortality in fishing gear (Ryan et al., 2012; Ryan and Ronconi, 2011). Although the nature of bycatch is fairly well understood, the link to population-level impacts has been harder to establish. However, a growing number of studies show negative relationships between fishing effort and adult survival or population trends (Rolland et al., 2010; Tuck et al., 2011; Véran et al., 2007). Assessing conservation implications (including critical areas and periods) requires estimation of bycatch rate or risk for each species in different fisheries based on the spatio-temporal overlap between fishing effort and bird distributions, as well as data on size and trends of affected populations (Small et al., 2013; Tuck, 2011; Tuck et al., 2011). Analyses need to consider not only bycatch by multiple fleets across ocean basins (Baker et al., 2007), including Illegal, Unreported and Unregulated fishing operations, but the impact relative to other threats (Rivalan et al., 2010; Rolland et al., 2010; Wanless et al., 2009). It is not necessarily the most frequently-captured species that suffer the most severe population-level consequences (Jimenez et al., 2012b). The Amsterdam albatross has a small but increasing global population, but models show that bycatch of only six individuals per year would eventually drive the species to extinction (Rivalan et al., 2010). Impacts of bycatch can also vary regionally; the impact on wandering albatrosses is much higher for breeding populations in the Atlantic than Indian Ocean (Poncet et al., 2006; Ryan et al., 2009; Tuck et al., 2011), whereas the reverse is true for white-chinned petrels (Ryan et al., 2012). Finally, bycatch can be biased towards males or females, potentially reflecting differential access to bait mediated by sexual size dimorphism, or sex-specific differences in foraging distributions (Bugoni et al., 2011; Nel et al., 2002a). This exacerbates the impact on breeding numbers by reducing effective population sizes and fecundity (Mills and Ryan, 2005).

5.2. Progress in mitigating threats from fisheries

A range of measures is available that can minimise bycatch, and improvements and novel approaches are still being researched. Although some approaches are widely-advocated, none is 100% effective in isolation. There is extensive variation in operational and gear characteristics among fisheries, and they may overlap with different assemblages of seabirds which vary in susceptibility to capture. Consequently, mitigation needs to be tailored carefully, and if introduced in combination with close monitoring of compliance has been very effective, for

example in trawl, demersal or pelagic longline fisheries around South Georgia, New Zealand, South Africa and Hawaii (Anderson et al., 2011; Bull, 2007, 2009; Croxall, 2008; Løkkeborg, 2011; Maree et al., 2014).

Mitigating seabird bycatch in pelagic longline is not as advanced as in demersal longline fisheries because of operational challenges to deploying bird-scaring lines, setting gear at night and attaching weighted swivels on branch-lines. Notwithstanding these difficulties, the efficacy of these approaches has been demonstrated through experimental studies, especially when used in combination, and without affecting target catch rates (Bull, 2009; Løkkeborg, 2011; Melvin et al., 2014; Robertson et al., 2013). In addition, although 'safe-leads' are available that reduce the risk of injuries to crew (Sullivan et al., 2012), there has been limited adoption by the pelagic longline industry (Baker pers. obs.). However, if appropriate mitigation is implemented, bycatch may be reduced significantly (Anderson et al., 2011; Gilman et al., 2014). Bycatch can also decline because of shifts or reductions in fishing effort, or changes in operational procedures that were not targeted specifically at bycatch reduction (Favero et al., 2013; Nel et al., 2002b; Robertson et al., 2014; Tuck et al., 2011). Best-practice bycatch mitigation has been adopted relatively recently by most tuna Regional Fisheries Management Organisations (trFMOs), but reductions in mortality can only be confirmed if there are vast improvements in observer coverage and data collection standards (see below).

Seabird mortalities associated with trawl fisheries are generally limited to the period when discarding is taking place (Favero et al., 2010; Maree et al., 2014; Pierre et al., 2012; Sullivan et al., 2006b). Therefore, avoiding release of discards while the warp cables are in the water would eliminate bycatch in most trawl fisheries. Complete retention of discards may not be operationally achievable, but management during shooting and hauling, and releasing batched waste at other times can reduce the attendance of seabirds, thereby mitigating associated risk (Pierre et al., 2012). The combination of improved discard management and the use of bird-scaring lines has reduced trawl bycatch significantly (Maree et al., 2014; Melvin et al., 2011; Pierre et al., 2012; Sullivan et al., 2006b). Efforts to address bycatch in gillnet fisheries are far less advanced, with very little concerted action to-date (Żydelis et al., 2013). Consequently, there is no current best-practice and an urgent need for further research.

ACAP routinely reviews bycatch mitigation measures and provides advice appropriate to each gear type. This advice needs to be complemented by increased awareness, education and training for operators, and appropriate regulations by management authorities. The Food and Agricultural Organization of the United Nations (FAO) has developed technical guidelines on reducing incidental catch of seabirds in capture fisheries that encourage adoption of National Plans of Action (NPOA-Seabirds) (FAO, 2008). To date, 14 states and other entities have formally adopted NPOA-Seabirds or their broad equivalent. In addition, BirdLife International and ACAP has jointly developed a series of fact sheets, available in several languages, which provide detailed information on each of the main mitigation measures, including technical specifications and implementation guidelines (<http://www.acap.aq/en/resources/bycatch-mitigation/mitigation-fact-sheets>). BirdLife International's Albatross Task Force have also achieved considerable success in building capacity on board vessels to refine mitigation measures (Croxall et al., 2012).

6. Threats: intentional take or killing at sea

Historically, albatrosses and petrels were deliberately caught at sea for human consumption, or shot from vessels for sport or scientific purposes (Robertson and Gales, 1998). More recently, intentional killing of seabirds to reduce the depredation of live bait in hook-and-line fisheries has been recorded off Brazil (Bugoni et al., 2008). Both incidental and intentional catches in Peruvian artisanal longline and gillnet fisheries are thought to have contributed to reduced adult survival, changes in sex ratios and population declines of waved albatrosses in the late 1990s

and early 2000s (Alfaro Shigueto et al., 2016; Anderson et al., 2008). Washing-up of broken wings provides circumstantial evidence for intentional take, although this may relate to the processing for food of bycaught birds, rather than active targeting. There is also circumstantial evidence from floating carcasses for intentional capture of black-browed albatrosses for food by squid-fishing vessels on the southern Patagonian Shelf (Reid et al., 2006). It is extremely difficult to quantify intentional take and its impact on populations, because the practise is likely to cease as soon as independent observers are on board. The factors underlying intentional take are different to those associated with bycatch, and require alternative solutions, including a greater focus on socio-economic and cultural issues (Alfaro Shigueto et al., 2016). Although it represents a less pervasive threat to albatrosses and petrels than bycatch, efforts need to be directed towards a better understanding of the contributing factors to allow effective conservation interventions.

7. Threats: pollution, debris and discarded fishing gear

As albatrosses and large petrels are long-lived top predators, they are potentially at high risk from bioaccumulation of marine pollutants through food chains. This applies even to species that feed in remote areas, as pollutants dispersed by long-range atmospheric transport continue to cycle in food webs for many years (Cossa et al., 2011; Nriagu and Pacyna, 1988; Riget et al., 2010). In addition, global emissions of mercury are predicted to increase, and although levels of some legacy (cf. emerging) persistent organic pollutants (POPs) are declining, there remains a high risk from new and emerging organic contaminants (Riget et al., 2010; Streets et al., 2009).

Concentrations of mercury, cadmium, arsenic and POPs in the tissues of albatrosses and petrels are related to trophic level, and also influenced by the degree of background contamination in foraging areas, and type of prey, including the proportion of squid, which have high cadmium levels in their digestive glands, and of mesopelagic taxa, which tend to have higher mercury burdens (Anderson et al., 2009; Anderson et al., 2010; Becker et al., 2002; Harwani et al., 2011; Stewart et al., 1999). There is evidence for increases in several pollutants, including mercury and organochlorines in the tissues of albatrosses in both the Northern and Southern hemispheres (Becker et al., 2002; Finkelstein et al., 2006). In black-footed albatrosses, these were associated with an alteration of immune function (Finkelstein et al., 2007), and the levels of PCBs and DDE were considered sufficient to increase the risk of eggshell thinning and reduce egg viability (Ludwig et al., 1998).

Mercury levels in adults and chicks of some species of albatrosses and large petrels can be well above the threshold associated with toxic impacts in terrestrial birds, yet cause no obvious deleterious effects (Blevin et al., 2013). This relates to the abilities to excrete mercury into feathers during moult, and into eggs by females, and in some species to demethylate mercury to its less toxic inorganic form (which can be sequestered in internal tissues); consequently, although mercury may increase in albatrosses from hatching to recruitment, the concentration then declines to a lower, stable level once adults have established a consistent moult pattern, and hence does not correlate with age in breeding adults (Tavares et al., 2013). Similarly, the toxicity of cadmium may be reduced by binding onto protein (metallothionein), and there is no evidence that cadmium concentrations increase with age to harmful levels (Stewart and Furness, 1998). By comparison, lead poisoning had an obvious deleterious effect on up to 5% of Laysan albatross chicks on Midway Atoll; however, this is an exceptional situation as the lead did not originate from prey but from ingestion of the paint used on old buildings in nesting areas (Finkelstein et al., 2003).

No published study suggests other than minor effects of oil spills on albatrosses or large petrels. Plastics have been found in their stomach contents, often mistaken for floating prey and ingested accidentally, including when scavenging behind fishing vessels, or, in the North Pacific

Ocean, ingested incidentally along with adhering egg masses from flying fish (Cherel and Klages, 1998; Fry et al., 1987; James and Stahl, 2000). Although in theory this may suppress appetite and partially or completely block the gut, there is little evidence for serious problems except possibly at the Hawaiian islands, where Laysan albatross chicks with high volumes of plastic in their proventriculus were significantly lighter at fledging (Sievert and Sileo, 1993). Plastics may become contaminated by toxic substances during manufacture, and floating plastic pellets in the marine environment adsorb toxic chemicals, including POPs (Colabuono et al., 2010; Mato et al., 2001). Plastic ingestion therefore increases the likelihood of contamination, particularly for chicks that tend to accumulate plastic particles in the gut until fledging. Albatrosses and large petrels are also at risk of ingesting discarded fishing gear, including hooks and line in offal, although the amount ingested shows substantial regional variation (Nel and Nel, 1999; Phillips et al., 2010; Ryan et al., 2016). A recent analysis of a 16-year dataset revealed that the amount of gear associated with wandering albatrosses was an order of magnitude higher than in other albatrosses and giant petrels, with a recent peak reflecting the adoption of a new longline system that resulted in greater discarding of hooks (Phillips et al., 2010). Despite the complete digestion of many hooks by chicks, fledging success remained high; however, whether toxic effects could be manifested after independence was unknown.

8. Threats: alien species at breeding sites

8.1. Impacts of alien species

Invasive alien species have had a destructive effect on wildlife worldwide, particularly birds and other fauna on islands which have not evolved effective natural defences against mammalian ground predators (Courchamp et al., 2003). The most widespread alien species with

Table 2

Number of breeding sites of albatrosses and large petrels (*Macronectes* and *Procellaria* spp.) that are affected by different levels of threat.

Nature of threat	Threat subcategory	Threat species	Number of breeding sites ^a affected		
			Threat level ^b		
			Low	High	All
Natural disaster	Sea-level rise	–	–	12	12
Contamination	Toxins – man made	–	1	–	1
Habitat loss or destruction	Habitat destruction by alien species	Reindeer	4	–	4
	Increased competition with native species	Australasian gannet	–	1	1
	Vegetation encroachment		2	–	2
Human disturbance	Military action		–	2	2
	Recreation/tourism		–	1	1
Pathogen	Pathogen	Avian pox virus	1	–	1
		Avian cholera	1	1	2
		Dog	–	1	1
Predation by alien species	Predation by alien species	Cat	11	2	13
		Pig	4	–	4
		House mouse	2	1	3
		Brown rat	6	–	6
		Black rat	9	–	9
		All		41	21

^a Breeding site: usually an entire, distinct island or islet, or rarely, section of a large island (>3000 km²). Each species-site combination is considered separately, i.e., two species breeding in the same area constitute two breeding sites. ACAP database. <data.acap.aq>. 14 July 2015.

^b See Appendix D for threat criteria.

the greatest impacts on seabirds tend to be predators, but invasive herbivores and plants can cause habitat deterioration, and introduced pathogens and insect vectors can become serious problems for animal health (Courchamp et al., 2003; Frenot et al., 2005). Of the mammalian predators, the most common threats to albatrosses and large petrels at breeding sites are feral cats *Felis catus*, brown rats *Rattus norvegicus* and black rats *Rattus rattus* (Table 2).

The impacts of invasive alien mammals are highly variable. There is evidence for predation of adult Laysan albatrosses by Polynesian rats *Rattus exulans*, several albatross and *Procellaria* petrel species by cats, royal albatross and Westland petrel chicks by stoats *Mustela erminea*, white-capped and light-mantled albatross, and Westland and black petrel chicks by feral pigs *Sus scrofa*, and adult and young Westland and black petrels by dogs *Canis lupus familiaris* (Croxall, 1991; Croxall et al., 1984; Kepler, 1967; Ratz et al., 1999; Taylor, 2000). Recent studies where the house mouse is the only introduced mammal have demonstrated predation on various albatross and petrel species at Marion and Gough islands (Cuthbert et al., 2013; Davies et al., 2016; Dilley et al., 2015; Dilley et al., 2013; Dilley et al., 2016; Wanless et al., 2009). Other introduced mammals that threaten ACAP species because of severe habitat degradation include pigs and reindeer *Rangifer tarandus* at a few sites (Table 2).

Population-level impacts of predation by alien species on albatrosses and large petrels are less common than might be anticipated. Although rat predation can cause widespread breeding failure in the burrow-nesting *Procellaria* petrels, no study has demonstrated a link between rat presence and population decline in the larger, surface-nesting albatrosses or giant petrels (Jones et al., 2008). In contrast, predation of Tristan albatross chicks by house mice at Gough Island is so common that this species, which is currently in rapid decline, would be unable to recover even if birds ceased to be killed in fisheries (Wanless et al., 2009). Although alien grazing mammals are present at several breeding sites of ACAP species, the associated habitat destruction appears only to have a substantial effect on distribution and, potentially, numbers of *Procellaria* petrels.

8.2. Progress in managing alien species

Given the major problems posed by alien species, there are ongoing management regimes aimed at local control of predators, including cats, mustelids or rats, at several breeding sites, including those of Westland petrel and royal albatrosses on the South Island of New Zealand, white-chinned petrels at Possession Island, and Laysan albatross in Hawaii (Taylor, 2000; Young et al., 2013). The number of high profile campaigns to eradicate alien mammals from islands is increasing, including nine past or ongoing eradications at breeding sites of ACAP species since the first ACAP Meeting of the Parties in 2004 (Appendix E). These include the successful campaign (10 years from planning to completion, at a cost of \$AUD 24 million) to eradicate European rabbits *Oryctolagus cuniculus*, black rats and house mice from Macquarie Island using a combination of rabbit calicivirus, aerial baiting, and hunting by a team with trained detector dogs. There has also been a three-phase campaign (baiting completed in March 2015) to eradicate brown rats and house mice from the 11,300 ha mainland of South Georgia, which if successful, would be by far the largest island ever cleared of rodents (Appendix E). It is important to recognise that these campaigns can result in substantial non-target mortality; >2500 birds died as a result of primary, secondary or tertiary ingestion of brodifacoum at Macquarie, including >760 northern and southern giant petrels, with substantial impacts on their local populations; however, non-target mortality was reduced by a range of mitigation measures, and it is anticipated that both populations will recover (Parks and Wildlife Service, 2014). Feasibility plans have also been produced for a number of other ACAP breeding sites, and in some cases planning is well advanced and eradications are scheduled for the next few years (Appendix E).

9. Threats: pathogens

The remoteness of their terrestrial breeding sites and their highly pelagic marine distributions likely shield albatrosses and large petrels from contact with pathogens in general. However, the associated immunological naivety may favour the rapid spread of pathogens should they be introduced to typically-dense breeding aggregations (Descamps et al., 2012), particularly if ongoing environmental changes increase the probability of establishment. Information on hosts, pathogens and disease epidemiology in ACAP species is incomplete, sampling is patchy in terms of geographic and species coverage, and very limited during the nonbreeding season, and there is a paucity of data on overall health and the ecological impacts of diseases. Potential pathogens have been recorded in 18 (62%) of the 29 albatrosses and large petrels (Uhart et al., 2014, Appendix C). Bacteria, viruses, protozoa, gastrointestinal parasites, ectoparasites and fungi were detected, respectively, in 7 (24%), 5 (17%), 4 (14%), 3 (10%), 13 (49%) and 1 species (3%). Seventeen different bacteria were recorded, most commonly avian cholera *Pasteurella multocida* (in four species) and *Salmonella* sp. (in two species). Only two viruses were isolated; pox viruses (in five species) and a new Phlebovirus (HIGV) in ticks from shy albatrosses. Recorded incidences reflect differences in research effort rather than environmental factors, with most studies focused on the black-browed albatross or southern giant petrel (16 and 15 papers, respectively).

The greatest risk appears to be from avian cholera, which is responsible for mortality events in several seabird species in Antarctica (Leotta et al., 2001; Leotta et al., 2003), and at Amsterdam Island, where it causes recurrent reproductive failure in Indian yellow-nosed and sooty albatrosses, and could potentially spread to the small population of the endemic, critically endangered Amsterdam albatross (Rolland et al., 2009). Amongst viruses, only poxviruses have been associated with disease or death, primarily in chicks or fledglings (five ACAP species, see Appendix C). Poxvirus outbreaks seem to be recurrent at some breeding sites, and sick birds often recover from the infection (Young and VanderWerf, 2008). Poxviruses and *P. multocida* are highly contagious and can be spread to remote locations by movements of animals, including scavenging birds, and human visitors. In terms of parasite infestations, only ticks and mites in black-browed and Laysan albatrosses, respectively, have been linked to disease or death (Uhart et al., 2014). However, this could change if ameliorating climatic conditions enable the establishment of insect vectors at higher latitudes.

10. Threats: climate change

An increasing number of studies in recent years have focused on potential impacts on seabirds, including ACAP species, of climatic variation, demonstrating effects of annual variation in sea surface temperature (SST) and marine productivity, and of global cycles (El Niño Southern Oscillation, North Atlantic Oscillation) (for reviews see Barbraud et al., 2011; Barbraud et al., 2012; Thomson et al., 2015). On land, warmer conditions can cause heat stress in chicks, and changes in rainfall and wind patterns can increase the risk of exposure. Higher SST, especially at foraging grounds, usually has negative effects on demographic parameters, especially breeding success, although the relationships can be non-linear. In contrast, black-browed albatrosses from Kerguelen benefited from increased SST, with evidence for contrasting responses to conditions in breeding vs non-breeding areas. Although juvenile survival can be reduced under warmer conditions, there is little evidence for a comparable effect on adult survival in albatrosses and petrels. Modelling suggests that responses to future climatic change will be species-specific, with few impacts predicted for northern species but steep declines for species in the Southern Ocean as a consequence of increased SST and decreased sea ice extent.

There have been shifts in distribution and breeding phenology of seabirds in response to climate change (Peron et al., 2010a; Weimerskirch et al., 2012). For example, changes in winds pattern

Table 3
Percentage of breeding sites and global population of each species of albatross and large petrel (*Macronectes* and *Procellaria* spp.) that are affected by terrestrial threats. Species without listed threats were excluded. See Appendix D for threat criteria.

Species	No. of sites	% of breeding sites ^a						% of global population							
		Natural disaster	Contamination	Human disturbance	Pathogen	Predation by alien species	Habitat loss or destruction by alien species	All threats	Natural disaster	Contamination	Human disturbance	Pathogen	Predation by alien species	Habitat loss or destruction by alien species	All threats
Antipodean albatross	6	0	0	0	0	17	0	17	0	0	0	0	1	0	1
Tristan albatross	1	0	0	0	0	100	0	100	0	0	0	0	100	0	100
Southern royal albatross	4	0	0	0	0	25	0	25	0	0	0	0	<1	0	<1
Wandering albatross	35	0	0	0	0	6	0	6	0	0	0	0	29	0	29
Short-tailed albatross	2	50	0	0	0	0	0	50	92	0	0	0	0	0	92
Laysan albatross	17	35	0	6	0	18	0	59	100	0	<1	0	<1	0	100
Black-footed albatross	15	47	7	7	0	7	13	60	98	34	0	0	0	38	98
Sooty albatross	15	0	0	0	7	7	0	14	0	0	0	3	12	0	15
Indian yellow-nosed albatross	6	0	0	0	17	0	0	17	0	0	0	69	0	0	69
Black-browed albatross	65	2	0	0	0	0	0	2	<1	0	0	0	0	0	<1
Shy albatross	3	0	0	0	33	0	33	66	0	0	0	67	0	2	69
White-capped albatross	5	0	0	0	0	20	0	20	0	0	0	0	6	0	6
White-chinned petrel	74	0	0	0	0	19	3	19	0	0	0	0	38	<1	38
Grey petrel	17	0	0	0	0	24	12	24	0	0	0	0	28	5	28
Southern giant petrel	119	1	0	0	0	0	0	1	?	0	0	0	0	0	?

^a Breeding site: usually an entire, distinct island or islet, or rarely, section of a large island (>3000 km²). Each species-site combination is considered separately, i.e., two species breeding in the same area constitute two breeding sites. ACAP database. <data.acap.aq>. 14 July 2015.

Table 4

Prioritisation of management interventions to address threats on islands with albatrosses and large petrels (*Macronectes* and *Procellaria* spp.). The prioritisation was only of threats at sites that hold >1% of the global population.

Island	Threat	Priority ^a	Explanation	Indicative cost (\$AUD) ^b
Habitat loss or destruction/predation by alien species				
Gough Island	House mouse	High	Major threat to endemic species; medium feasibility of eradication	5.5 million
Grande Terre, Kerguelen	Reindeer	Lower	High feasibility of eradication	1–2 million
	Feral cat	Lower	Medium feasibility of eradication	>10 million
	Black rat	Lower	Medium feasibility of eradication	>25 million
Ile Saint Lanne Gramont, Kerguelen	Feral cat	Lower	High feasibility of eradication	420 K
	Black rat	Lower	High feasibility of eradication	140 K
South Georgia (Islas Georgias del Sur)	Brown rat ^c	Lower	Medium feasibility of eradication	15 million
Auckland Island	Feral cat	Lower	Medium feasibility of eradication	25 million
	Domestic pig	Lower	Medium feasibility of eradication	25 million
Marion Island	House mouse	Lower	Medium feasibility of eradication	30 million
Pathogen				
Ile Amsterdam	Avian cholera	High	Major threat to two species; low or unknown feasibility of eradication	Unknown
Increased competition with native species				
Pedra Branca	Australasian gannet	Lower	Low or unknown feasibility of eradication	100 K

^a High priority reflects major threat to an endemic species or very large proportion of the global population.

^b Economies of scale would reduce overall cost of operations in same island group.

^c Aerial poison baiting completed (2015).

have modified the distribution of wandering albatrosses in the Indian Ocean, and resulted in improved body condition and breeding success. Other impacts of climate change that may be deleterious are changes to weather, including rainfall patterns, that could lead to increased surface erosion and loss of nesting habitat because of landslips (Ryan, 1993). Sea level rise is also likely to increase susceptibility of albatross colonies on low atoll islands in the Pacific Ocean to submersion during storm events (Storlazzi et al., 2013). Warming conditions might also lead to a potential increase in risk of transmission of diseases because of greater nutritional or environmental stress in infected birds, and increasing abundance or the establishment of new vectors. Apart from the obvious global interest in minimizing climate change by reducing greenhouse gas emissions, direct impacts on land may be reduced by improving habitat management to reduce erosion, or establishment of new colonies at suitable sites by translocation or attracting recruits using decoys or tape playback (Deguchi et al., 2014).

11. Threat prioritisation

ACAP has adopted standardised, objective systems for the assessment of threats to albatrosses and petrels, both at sea and on land (Appendix D). On land, the threats affecting the greatest number and proportion of breeding sites, and proportion of the global population of each species, relate to habitat destruction and predation by introduced mammals, although some other threats present at just a few sites are severe (Tables 2 and 3). The two species affected at the most breeding sites are grey petrel and white-chinned petrel, which are burrow nesting, mainly because of predation or habitat destruction by introduced mammals (Table 3). Management interventions that would remove threats were prioritised based on a score that combined vulnerability (reflecting global population size, proportion of global population and population trend at the site), threat magnitude, and likelihood of success (Table 4). The analysis was only of important global breeding sites (>1% of the global population; see Appendix A), and scores for threats that applied to more than one species in the same area were summed. On this basis, by far the two highest priorities were on islands where there was a major threat to an endemic species or very large proportion of the global population; to eradicate house mice from Gough Island and to mitigate impacts of avian cholera at Ile Amsterdam. The scores for the other threats from alien species all differed from each other by ≤ 2 , and were therefore in a large group considered to be Lower priority. Indicative costs are provided in Table 4 based on expert opinion, but were not used in the prioritisation process. The bulk of the costs are associated with planning and mobilisation, and

hence economies of scale would be substantial if an eradication campaign targeted more than one species at the same island or island group. In most cases, there would also be value in removing introduced vertebrates from islands that were formerly occupied or stand a good chance of being colonised by species of conservation concern (Rauzon, 2007; Towns and Broome, 2003).

ACAP has also developed a framework for the assessment and prioritisation of at-sea (fisheries) threats. Currently, a total of 87 fisheries-seabird population combinations is identified as being of high priority for conservation action. However, many of the fisheries affect multiple seabird species and populations, and the combined list of priorities includes 28 seabird populations and 27 fisheries (Appendix F).

12. Future challenges for albatross and petrel conservation

Despite considerable improvements in recent decades in knowledge of ecology, distribution, population sizes and demography of albatrosses and large petrels, many gaps remain. These gaps include information on population size, trends and threats at major breeding sites, and on at-sea distributions and levels of interaction with fisheries of immature birds, and of adults during the nonbreeding season. Although conservation management has been better targeted in recent years, these species still face a wide range of often very serious threats in marine and terrestrial environments. To address the most pervasive threat - bycatch - will require wider and more effective implementation and, in some cases, further development of best-practice mitigation measures in national (particularly gillnet, trawl and artisanal) and international fisheries (particularly pelagic longline), and much better information on bycatch rates and levels of compliance. More research is required on the effects of introduced vertebrates on burrow-nesting petrels and other less easily-observed species. Although there have been successful, high-profile eradications of alien species from islands in recent years, and further campaigns are planned or warranted, there remains a need for better representation of the underlying science in the peer-reviewed literature in order to improve methodologies, reduce risk of failure, and minimise the poisoning of non-target species (Phillips, 2010). Other threats that require more research to better understand current effects and predict future impacts include those from oceanographic and other changes in the wider ecosystem (requiring more data on diet, distribution and demography), infectious diseases (including the establishment of systematic monitoring to determine baseline occurrence of pathogenic organisms) and pollutants. Allocating more resources to research and to advocating for improved management and monitoring of fisheries

and other threats may provide the only means of securing a positive future for albatrosses and large petrels.

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Appendix. Supplementary data

Supplementary data to this article can be found online at <http://dx.doi.org/10.1016/j.biocon.2016.06.017>.

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